Chapter 1. Introduction

1.1 The Feedlot System

A cattle feedlot is an intensive agricultural system, where cattle are fed specific rations in a confined area, generally for several months. The rations are formulated and the time cattle are retained are such that the end product meets specifications required by different markets. These markets range from the domestic to international markets, such as Korea and Japan. Due to the design of the feedlot production system, a continual input of animals and rations occurs, which result in the output of manure and effluent containing varying amounts of nutrients. These nutrients become inputs to storage and crop production systems and potentially can degrade the external environment if released into it. This study focuses on the development of database and modelling tools to enhance the understanding of the nutrient pathways in and from the crop production system.

Nutrients contained in manure and effluent cycle through the whole feedlot system. The feedlot can be broken down into several sub systems, with the central component of a feedlot being the animal feeding system, where animals are fed and housed in confined areas at a density of 10-25 m^2/head (SCARM 1997). Manure and effluent are by-products of the feeding operation and have been seen as either a waste to be disposed or a valuable resource to be utilised. The design and management of the feeding operation and manure and effluent storage facilities all contribute to the characteristics of these by-product resources.

An ideal nutrient pathway in a system utilising manure and effluent, would be one where the nutrients are totally contained in the animal, soil or plant systems and do not migrate to the external ground water or surface water systems. Ideal nutrient pathways for the nutrient cycling in a typical feedlot are shown in Figure 1.1. In reality, the system is not closed and the nutrients can and do reach the external environment. While these events are undesirable, it is not practical to completely eliminate them and management strategies should be designed to minimise nutrient loses via leachate and runoff and maximise crop uptake.

Feeding and storage components of the feedlot have been extensively researched, providing a knowledge base for the design and management of animal production systems and manure and effluent storage. However, these same advances have not been parallelled in the manure and effluent utilisation area. Monitoring each sub-system and the external environment should provide information necessary to advance the knowledge of the important processes and interactions in the utilisation area.

The word utilisation is used exclusively in this study when referring to the application of manure and effluent that is accumulated as a result of the intensive housing of cattle for the purposes of beef production. This is deliberate, as the concept of having to dispose of a waste as opposed to the utilisation of a resource encompasses a totally different viewpoint in the context of managing the system. It would seem more likely that when the manure and effluent are viewed as a fertiliser and irrigation source, the nutrients contained within them would be applied and managed in such a way as to obtain the optimum production system.

Utilising manure and effluent can be an asset to the feedlot production system, whilst having the potential to be a liability to its own and the surrounding environment. To maximise the asset and minimise the liability firstly requires that the inputs, changes of state and outputs of the crop utilisation system be defined. Monitoring these aspects is the next requirement to minimise the liability to the environment and the final step is to make use of the information gathered for the optimal management of the system. Monitoring data should be used as a means of developing an understanding of the important processes that can be manipulated to produce an optimal production system and an environment that is not degraded.

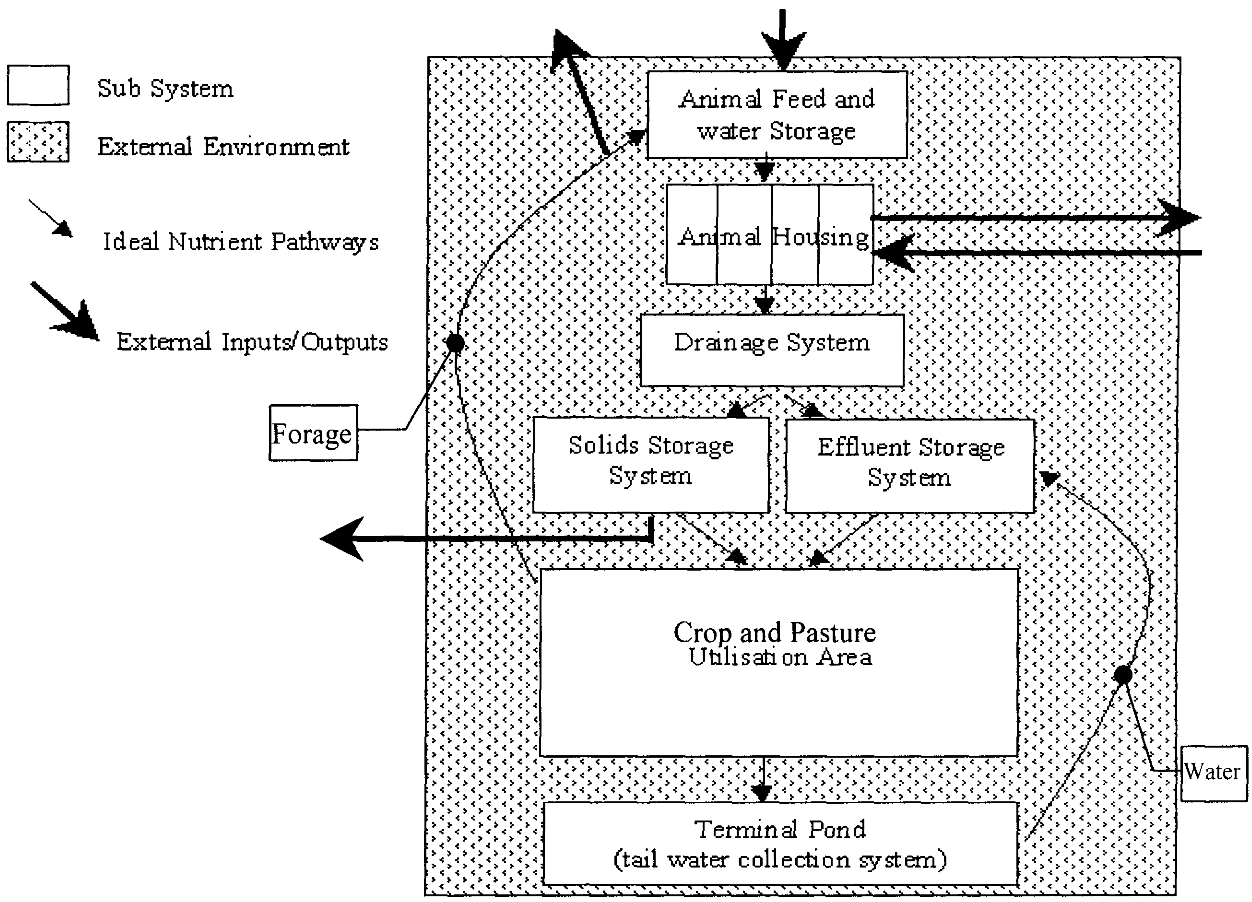

Figure 1.1. An Example of a Feedlot Layout and Nutrient Pathways (Assuming a Closed System)

1.2 Monitoring the System

Because of the potential for nutrients to have detrimental effects on the feedlot and its surrounds, Australian regulatory authorities require that feedlots monitor several aspects of their environment. These requirements vary from state to state and are dependent on the size, age and location of the feedlot. Regardless of the legislative requirements, feedlots should monitor inputs (feed rations, stock water, fertiliser etc.), outputs (crops, manure and effluent etc.) and the changes that are occurring in the soil, ground water and surrounding surface waters. This monitoring should focus not only on environmental aspects, but also on providing feedback for the optimal management of the production system. Data should be obtained from the soils, air and ground water above, within and below the feedlot facility, as well as surface water upstream and downstream of the facility. These data, along with effluent and manure samples should be collected on a regular basis. Data also needs to be obtained from the soils and crops within the utilisation area.

1.3 Thesis Aim and Objectives

The aim of this thesis is to develop an understanding of the important nutrient pathways when cattle feedlot manure and effluent are used as inputs to a crop production system. This understanding is achieved through the investigation of the cycling of nutrients within and movement of nutrients from an irrigation area. Nutrients included in the study are nitrogen (N), phosphorus (P), sodium (Na), potassium (K), calcium (Ca), and magnesium (Mg).

This study draws primarily on research carried out at the Cooperative Research Centre for the Beef and Cattle Industry feedlot facility 'Tullimba'. Tullimba is a 1000 head research feedlot, which has extensive monitoring programs in place for all sub-systems and the external environment (Figure 1.1). Data have been intensively collected as part of the research associated with the nutrient, salt and water balances within the feedlot. These data have been utilised within this thesis, where a database was built to store and manipulate data. The database was extended by including a stochastic model focusing on comparing the effects of different application rates and time of application of manure and effluent to predict the magnitudes of each of the pathways.

1.4 The Research Approach

A review of the literature is carried out to identify the components required in a model to investigate the sustainability of a system utilising manure and effluent. These components are:

- representation of the stochastic nature of the system;
- cation exchange processes;
- N and P transformations;
- soil water distribution;
- an account of the flux of the nutrients through the output pathways of the system (crop uptake, leachate and runoff); and
- probability distributions of the output.

All of these components are included in the model developed to carry out the research for this study. The model is also developed with a view to using data collected as part of the normal monitoring requirements of a beef cattle feedlot. This is to overcome the problem of transferring an agro-ecosystem model developed at the laboratory scale to the field scale (Gaunt *et al.* 1997; Mirschel *et al.* 1997). The aim of developing the model in conjunction with the database is to allow seamless integration of input data into the model.

There are four main objectives required to achieve this aim, each with identifiable goals, as outlined below.

1. Develop an Environmental Monitoring Database (EMD) to provide (Chapter 4):
   - an efficient and secure data storage system with easy data entry and retrieval; and
   - an interface for investigating data to highlight the important processes and interactions within, and external to, the whole system.
2. Develop a simple, robust, conceptual model of the utilisation of manure and effluent to investigate the output pathways of N, P, Na, K, Ca and Mg (Chapters 5 and 6). The important components to include in this model are:
   - the variability of the feedlot manure and effluent utilisation system;
   - the stochastic nature of this system as a characteristic of the model;
   - use of data collected for monitoring purposes as model input;
   - a predictive capability and a measure of sustainability in the form of ratios of outputs to inputs; and
   - a seamless integration of input data, using Microsoft Access as the modelling interface.
3. Evaluation of the model performance with consideration to (Chapter 7):
   - the simple mass balance approach used in the model development; and
   - available datasets.
4. Apply the model to the Tullimba irrigation area and use the output to predict the performance of the system (Chapter 8). The aims of undertaking this modelling of the Tullimba irrigation area are to:
   - observe the predicted changes in the cation balance in the soil, as a result of the manure additions;
   - demonstrate the use of the model to run virtual experiments with different manure application rates;
   - obtain probability distributions of the amounts of N, P, Na, K, Ca and Mg removed in the runoff and leachate on a monthly basis; and
   - identify the critical months when predicted losses via runoff and leachate are likely to be greatest.

Chapter 2. Sustainability and Background Studies

2.1 Defining Sustainability

Environmental sustainability is linked to the concept of Ecologically Sustainable Development (ESD) (Lott *et al.* 1996), a concept that has gained acceptance in recent times. The Department of Environment and Heritage (1992) cited a definition for ESD which arose from an Australian Government report on the subject: 'using, conserving and enhancing the community's resources so that ecological processes, on which life depends, are maintained, and the total quality of life, now and in the future, can be increased. Put more simply, ESD is development which aims to meet the needs of Australians today, while conserving our ecosystems for the benefit of future generations.' A similar definition to the one given above was supplied by Lee and Pankhurst (1992), who suggest it is widely accepted in Australia that conserving our existing resources and trying to repair previous damage to land and water resources is a general principle of land use.

In part, the driving force behind the growing awareness of ESD principles has been society's concerns about the degradation of natural resources and the environment (Edwards *et al.* 1993; Hamblin 1991; Kruseman *et al.* 1996). There is also an increasing awareness by agricultural producers of the responsibilities they have towards the environment, and an understanding of the effects of their operations on the environment (Duggin & Murray 1997; Park & Seaton 1996; Sriskandarajah & Dignam 1992).

Sustainable agriculture as a concept was first mooted in the 1980s (Edwards *et al.* 1993) and there is increasing pressure to formulate a framework that can be used to quantify sustainable agricultural systems (Hansen 1996; Kruseman *et al.* 1996; Park & Seaton 1996; Stockle *et al.* 1994). However, a search of the literature reveals many complex aspects that have been described as essential to incorporate into a measure of sustainability. The situation is further complicated by the concepts relating to sustainability that are undergoing changes and shifts in emphasis (Kruseman *et al.* 1996), with the emergence of some broader issues of sustainability (Sriskandarajah & Dignam 1992).

The components of sustainability identified by the recent literature include environmental/ecological impacts (Blaschke *et al.* 1992; Chandre Gowda & Jayaramaiah 1998; Kruseman *et al.* 1996; Park & Seaton 1996; Sriskandarajah & Dignam 1992), inter and intra generation considerations (Kruseman *et al.* 1996; Park & Seaton 1996), the production system (Blaschke *et al.* 1992; Chandre Gowda & Jayaramaiah 1998; Hansen 1996; Kruseman *et al.* 1996; Sriskandarajah & Dignam 1992), social, political and legal constraints (Ison *et al.* 1997; Kruseman *et al.* 1996; Park & Seaton 1996; Sriskandarajah & Dignam 1992) and economic considerations (Kruseman *et al.* 1996; Park & Seaton 1996). A dynamic systems approach is also considered by many authors (Hansen 1996; Hansen & Jones 1996; Ison *et al.* 1997; Kruseman *et al.* 1996; Sriskandarajah & Dignam 1992).

To characterise sustainability, the common elements include spatial and temporal scales and an ability to continue into the future (Blaschke *et al.* 1992; Edwards *et al.* 1993; Hamblin 1991; Hansen 1996; Hansen & Jones 1996; Kruseman *et al.* 1996; Park & Seaton 1996; Stockle *et al.* 1994). A hierarchy of levels and an interconnectivity of systems are also considered by some authors as being important in characterising sustainability (Edwards *et al.* 1993; Kruseman *et al.* 1996; MRC 1996; Park & Seaton 1996; Stockle *et al.* 1994).

The criteria and measurements most often applied to sustainability are operational (Kruseman *et al.* 1996), qualitative and quantitative (Chandre Gowda & Jayaramaiah 1998; Park & Seaton 1996) and consider the integrated nature of the system (Hansen 1996; Park & Seaton 1996). Conceptual frameworks and semantic descriptions are also offered as a means of specifying the criteria of sustainability (Kruseman *et al.* 1996).

Aggregate measures, rate and state variables, and interactions between processes and indicators (Kruseman *et al.* 1996) are all used as measures of the resource base and system performance, such as long-term yield projections (Stockle *et al.* 1994) and measures of external effects (Hansen 1996; Kruseman *et al.* 1996). These measures capture the adaptability, flexibility and resilience of the system, and are important for quantifying sustainability (Kruseman *et al.* 1996; Park & Seaton 1996). However, the most difficult aspect to quantify is the system's ability to continue into the future.

Sustainability, in the context of continuing into the future, is applicable to dynamic systems that are continually changing and developing (Hansen & Jones 1996). Agricultural development implies a change to the ecological system and therefore the original ecosystem is not the system to be sustained. For this reason, the following discussion will focus on management practices for a sustainable agricultural production system that utilises manure and effluent resources produced through intensive animal farming. By its very nature, this system is required to be adaptive to change.

2.2 Defining an Agro-ecosystem

Edwards *et al.* (1993) defined an agro-ecosystem as an integrated farm model that considers the whole system and the socio-economic and biophysical flows within and through the system. An ecological system that has been modified by humans for the purpose of producing food, fibre, or other agricultural products is another definition provided by Blaschke *et al.* (1992). Agricultural science and ecology both contribute to the knowledge required to understand the functions of, and processes occurring within the agro-ecosystem (Edwards *et al.* 1993).

The characteristics of an agro-ecosystem include (Frissel 1978):

- an organisation of resources, managed to a greater or lesser extent by humans;
- a system with production of human food and fibre as one of its main objectives;
- being frequently characterised by the input of fertiliser and the output of food or fibre products; and

- being especially subject to influence by humans and their cultural operations.

These characteristics are applicable to a system that utilises manure and effluent produced by a feedlot.

2.2.1 The Feedlot Agro-ecosystem

The basic principles applying to a conventional system will still be relevant in the context of the utilisation of the resources produced by a feedlot. However, the added dimensions and complexities of the increased organic matter, nutrient concentrations and the combination of nutrients and salts being added to the system simultaneously need to be considered.

Agro-ecosystems of the past were integrated, in that the crop and animal production units were symbiotic (Azevedo & Stout 1974) and the move away from these production systems can be attributed to several reasons (Azevedo & Stout 1974; Norstadt *et al.* 1977; Sander 1996; Sri Ranjan *et al.* 1995; USDA & US EPA 1998):

- agricultural producers specialising in one or a narrow set of commodities, rather than mixed farming; and
- the shift from viewing manure as a fertiliser to that of a material of comparatively little value, and therefore a waste product.

Some authors see the design and management of practical integrated systems of crops and animals as the challenge that needs to be met for a sustainable system (Edwards *et al.* 1993). With the advent of feedlots, the bulk of the animal feed is produced in a location remote from where the manure is deposited (Azevedo & Stout 1974). Therefore, the nutrient cycle of the feedlot as a whole has become unbalanced, with the inputs being increased while at the same time decreasing output pathways. An understanding of the system from a holistic perceptive will enhance the development and management of an integrated agro-ecosystem.

2.3 Background to the Elements of Sustainability of Agro-ecosystems Relevant to this Study

Sustainable agro-ecosystems can be classified according to the motivating issues of concern. Several classifications provided by Hansen (1996) are outlined below.

- Sustainability as an ideology. This embraces a philosophy and a system of farming that integrates land stewardship with agriculture. Development is in response to concerns of the negative impacts of agriculture, with the underlying goal of motivating the adoption of alternative practices.
- Sustainability as a set of strategies. These include the reduction of inputs, minimisation of the impact of the system on the immediate and off-farm environment, a sustained level of production and financial profit, maintaining or enhancing environmental quality, providing adequate economic and social reward to all individuals in the production system and production of sufficient and accessible food supplies. Such goals must endeavour to achieve a greater efficiency of resource use, and not degrade the ecosystem.
- Sustainability as an ability to continue. To maintain productivity in spite of a major disturbance, such as is caused by intensive stress or a large perturbation, which incorporates a predictive

property into sustainability. This is a key element of sustainability, and requires the quantification of the system inputs and outputs.

The methods of achieving sustainability are dynamic and the known constraints to sustainability change in tandem with technological advances and the introduction of improved management practices. This aspect makes it difficult to apply, in a practical way, most of the definitions for sustainability from the recent literature (Blaschke *et al.* 1992; Ison *et al.* 1997). Characterising sustainability is necessary to apply the three concepts of sustainability listed above, which requires a literal interpretation and a system-oriented approach (Hansen 1996). A quantitative approach is also required to enable comparisons between agricultural systems (Kruseman *et al.* 1996).

Quantitative indices include measures of profitability (income and credit ratings), productivity (yield measurements), soil (erosion rates, organic matter content, cation exchange capacity, salinity, alkalinity, infiltration, water holding capacity and earthworm activity), water (leachate and surface water quality) and air quality (odour intensity and particulate matter concentrations), energy efficiency (fossil fuel dependence, output/input energy ratio), wildlife habitats, quality of life and social acceptance (Hansen 1996; Stockle *et al.* 1994). Obviously there are many practical difficulties integrating all these aspects into a single measure of sustainability, but a quantitative measure of sustainability should include a probability of continuing into the future along desirable and viable pathways (Hansen 1996).

Park and Seaton (1996) presented a framework for agro-ecosystem research in which sustainable pathways are described in terms of desirability and viability space. Any pathways that transverse both "spaces" are said to be economically beneficial in the short-term and ecologically sound over longer time periods. The progression along these sustainable pathways 'requires an understanding of the vertical and horizontal integration between and within a variety of production systems that interact with the natural environment' (Park & Seaton 1996).

The viability and sustainability space framework upon which Park and Seaton's (1996) sustainable model is built incorporates changes with respect to time. The viability space encompasses the shorter-term objectives such as the maintenance of income, employment, capital investment etc. The larger desirable (or sustainable) space of the long-term objectives incorporates such things as nutrient balances, water quality and adaptable systems. The pathways of development and adaptation must pass through both these viability spaces for the system to be sustainable. In essence, the viability and sustainability spaces are mapped onto each other, as shown in Figure 2.1.

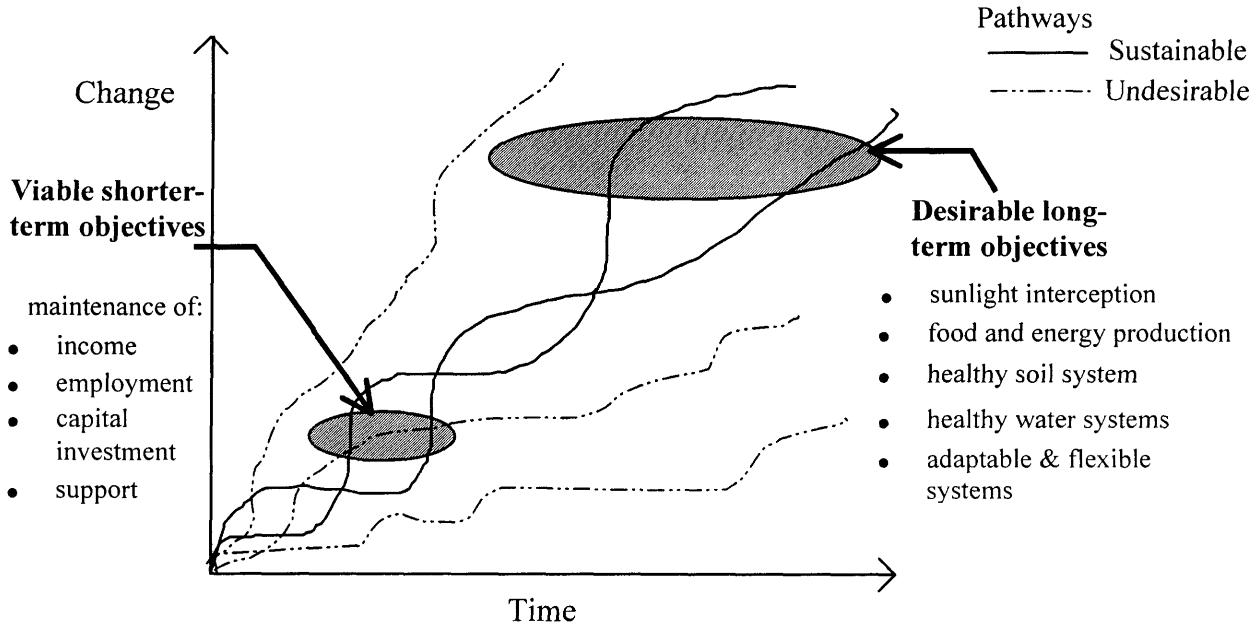

Figure 2.1. Sustainable Pathways and Viability Space as a Function of Time (Adapted from Park & Seaton 1996)

A simulation model can be used to provide some insight into the nature of these sustainable pathways by producing a set of replicated time paths through the random sampling of stochastic variables driving the system (Hansen & Jones 1996). The objective of such a model should assess sustainable pathways in relative terms and not necessarily produce predicted values that can be matched exactly with measured values, for example, the pathways associated with different management practices. There is however, a lack of methods available for designing and interpreting simulation studies used to characterise sustainable systems (Hansen 1996).

Two sustainable conditions were the basis of a model for the design and management of an effluent irrigation scheme provided by Hu (1997). The first sustainable condition is such that the accumulated amount of any pollutant is controlled below a value that the soil can assimilate without damaging the land. Sustainable condition 2 is such that the amount leaving the system via runoff is below the acceptable levels required by the governing rules of the water bodies within the catchment.

Six pollutant species were considered in Hu's (1997) sustainable irrigation model. A mass balance of each of these species is used to obtain the amount lost via runoff and the amount stored in the system. A sustainable index is then used to provide a measure between 0 and 1. Figure 2.2 is a graphical representation of the index. When the sustainable index equals 1 and the measure of the pollutant equals 0, the system is sustainable. At some measure of the pollutant load a point called C_{safe} is defined, which is a threshold value above which the pollutant has the potential to produce adverse effects. From C_{safe} there is a decreasing linear relationship between the sustainable index and the increasing pollutant load. When the sustainable index equals 0, another point, C_{damage} is defined. At this point any damage to the catchment or area cannot be recovered. The sustainable index for the whole system is the minimum of the sustainable indexes for each pollutant species considered.

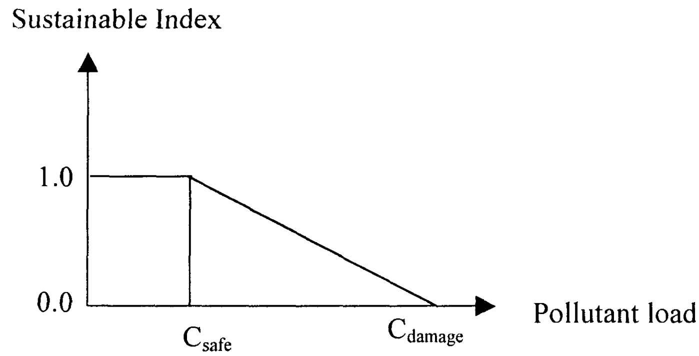

Figure 2.2. Sustainable Index (Adapted from Hu (1997))

Hu (1997) defined the sustainable design of an effluent irrigation scheme from the perspective of a disposal problem and not an opportunity to utilise a resource. Further, the model used by Hu (1997) to produce the sustainable index is empirical and does not account for the stochastic nature of any of the variables that influence the response of the system. The relationship between the sustainable index and the pollutant load is assumed to be linear, which may not always be the case. There also does not appear to be any provision in the model for investigating the effects of different management practices. Interactions of the "pollutants" are also not investigated.

2.4 Quantifying Sustainable Agro-ecosystems

There is no single, summary indicator that will explicitly quantify sustainability (Stockle *et al.* 1994) and some of the criteria found in the literature for use within a sustainable systems framework are ill-defined and not practical to apply (Park & Seaton 1996). This is due to the difficulties associated with applying a quantitative and predictive measure as a central component of sustainability. However, the first step in developing a measure of sustainability is understanding the system interactions and processes that occur and the temporal scales that most affect this measure.

Once an understanding of the system has been reached, monitoring systems should be developed to provide quantitative information, which in turn provides the basis for developing management guidelines for sustainable land use (Blaschke *et al.* 1992). Monitoring the system is the second step in determining the sustainability status of a system, but this is a long-term endeavour. In most cases, research into the important system interactions and processes is required over several or more years before monitoring can provide any practical indications (Stockle *et al.* 1994).

Because of the long-term nature of monitoring, simulation models are recognised by many authors as a necessary component in the framework for evaluating the sustainability of a particular agro-ecosystem (Chandre Gowda & Jayaramaiah 1998; Hamblin 1991; Hansen 1996; Hansen & Jones 1996; Park & Seaton 1996; Stockle *et al.* 1994). This can be achieved by using a simulation model to investigate a range of scenarios. Important components of the simulation model are a baseline measure of the agro-ecosystem (Stockle *et al.* 1994) and a suitable proxy measure of change to provide reliable feedback about future impacts of current decisions (Hansen & Jones 1996; Park & Seaton 1996). Another important component to capture in any baseline measure is the spatial and temporal heterogeneity inherent in the system being modelled.

Four categories are recognised for grouping indicators of sustainability (Hamblin 1995; Hansen 1996):

- management indicators;
- production indicators of system performance;
- indicators of effects on other systems; and
- indicators of the resource base status.

The economic, planning and operational aspects of the system are included in the management group. The production indicators are based on ratios of crop productivity relative to inputs of water, nutrients, labour and other variables. These production indicators include nutrient balances expressed as ratios of nutrients removed in products to nutrient uptake and the proportion of applied nutrients versus nutrients and sediments entering rivers and ground water (Hamblin 1995). The resource base status indicators are the biological health, fertility and physical condition of the soil, and the condition of the surface water and ground water resources.

Indirect indicators of change can be used in conjunction with models to make the model output more reliable. Bio-indicators are used to monitor soils and wider ecological change (Park & Seaton 1996). An example is the use of the mass of soil invertebrates as a simple proxy measure of agro-ecosystem health. Soil animals can act as a sensitive indicator of the state of soils and the impacts of environmental changes (Park & Seaton 1996).

2.4.1 The Predictive Components of Sustainability

Hansen and Jones (1996) restated the definition of sustainability as 'the probability that a particular system will not meet specified criteria for failure during a particular future period'. Because sustainability deals with future changes, it must be predictive of the future rather than simply descriptive of the past or present (Hansen 1996). A dynamic stochastic system is, by it very nature, a system that is characterised by its variability (Hansen & Jones 1996) and a stochastic approach would seem necessary in order to express predictions as probabilities (Hansen 1996). Therefore, the essential components of any measure of sustainability are a predictive capability within a systems approach.

System attributes that influence sustainability are measures of the resource base, system performance and effects on other systems (Hansen 1996). A simulation model can be used to produce these measurements and observe their relationships with other key indicators that are known to be problematic, such as nitrate leaching and P in surface runoff. The incorporation of baseline data into the resource base, system performance and effects on other system measures will provide a platform from which to observe trends.

The measures of the resource base must highlight those variables that are particularly prone to changes in the natural resource stocks, such as soil nutrients (Hansen 1996), particularly soluble nutrients. An agro-ecosystem is not as efficient as a mature ecosystem in trapping and holding nutrients, and the nutrients have the potential to leak readily from the system (Edwards *et al.* 1993). Therefore, an important determinant of sustainability is the soil nutrient flows and this requires monitoring and simulation of the soluble and exchangeable components of the elements of interest.

Another important determinant of sustainability is the expectation of long-term yield levels, which is often quoted as an indicator central to any sustainable system (Park & Seaton 1996; Stockle *et al.* 1994). Yield levels are dynamic and reflect the current state of knowledge, technical advances and economic, social and ecological conditions (Park & Seaton 1996). For example, a simulation model can be used to predict a sustainable system based on the system's ability to produce a potential yield in 20 years time. However, in 20 years time, technological changes in agriculture may mean that a far higher yield level is seen as sustainable. Therefore, any simulation model used for predicting sustainability should be used dynamically and on a regular basis. Further, inputs should reflect the current state of knowledge and the output should be characteristic of the system's behaviour.

The system's central tendency and variability (Hamblin 1991; Hansen & Jones 1996; Stockle et al. 1994) influence the behaviour of a system; all of which can be captured in a simulation model that uses Monte Carlo techniques (randomly sampling a probability distribution). An extremely accurate model may not be necessary if the relative sustainability of different management practices (Stockle *et al.* 1994) and an indication of the broadly desirable/undesirable temporal pathways (Park & Seaton 1996), is all that is required. If this is the requirement, then modelled output within the same order of magnitude of measured data should suffice.

Diagnosing the cause of unsustainable practices and predicting the effects of these practices, is an important capability of a simulation model (Hansen 1996). For example, the effects of leaching on acidification of the soil. If the ion exchange reservoir of the soil is allowed to diminish to the point where it is nearly empty, then recovery may be in the order of centuries (Sverdrup *et al.* 1995). Therefore, waiting until damage is evident on the surface would be catastrophic.

When a simulation model is used to obtain a general trend, the output required is a vector of development indicators that are updated over time. If there are positive changes in the vector values over time for each and every time period, the sustainability is classified as 'strong'; if there is only a positive trend in the values then the sustainability is defined as 'weak' (Kruseman *et al.* 1996). Negative changes imply an unsustainable system.

Another important tool required for the ongoing measurement and understanding of sustainability is a decision support system that integrates expert opinions and dynamic simulation models (Ison *et al.* 1997; Park & Seaton 1996; Stockle *et al.* 1994). The decision support system and model should be used to provide a dynamic measure of sustainability, with real data continually input into the model so that a general trend can be established. When a decision support system is provided as a tool for evaluating the sustainability of an agro-ecosystem, its objective in this regard, must be clearly stated for the tool to be used correctly.

2.5 Comprehensive Nutrient Management Plans (CNMPs)

A confined animal feeding operation (CAFO) is defined as an agricultural enterprise where animals are kept and raised in confined situations (USDA & US EPA 1998). These animals and production operations are concentrated on a small land area, with feed brought to the animals as opposed to the animals grazing on pastures, and in 1968, more than 67% of American beef came from such operations (Azevedo & Stout 1974). Current estimates suggest that 450,000 agricultural operations in the US confine animals (USDA & US EPA 1998). However, the move to this type of beef production did not occur in Australia until the 1970s and serious expansion did not occur until 1988 (Bindon 1996).

The USDA Draft Strategy for Animal Feeding Operations requires the implementation of technically sound and economically feasible Comprehensive Nutrient Management Plans (CNMPs) (USDA & US EPA 1998). These plans should be designed so that the CAFO has no adverse impacts on water and soil quality and public health. The basis of the CNMP is its role in identifying actions or priorities needed to meet the defined nutrient management goals of a CAFO.

The CNMP should encourage and facilitate technical innovation and new approaches to manure and nutrient management. The ultimate responsibility for the development and implementation of the CNMP is the CAFO operator (USDA & US EPA 1998). Thompson *et al.* (1997) reviewed a range of decision support systems available for assisting feedlot operators to meet the requirements of the CNMP by calculating optimum manure application rates. However, these decision support systems are only concerned with the utilisation of solid manures and do not include any aspects of effluent irrigation.

Six components of a comprehensive nutrient management plan are outlined by the USDA draft strategy for animal feeding operations (USDA & US EPA 1998). The first component is aimed at producing manure that has a N:P ratio close to that required by the plant through the adjustment of the feed content or the addition of enzymes that increase the utilisation of P by the animal. However, this ratio can be difficult to control when rations have been 'bulked out' with salts by the manufacturer.

Other elements in manure and effluent are of concern, particular sodium (Na) and aluminium (Al). Na is a concern because of its known ability to reduce yields through increasing the osmotic potential within the soil solution. Na is also easily leached from the soil and because the plant does not readily take it up, the

Na is transported through to the ground water. Although the absolute amount of Al applied is small compared to the inputs of some of the other elements, the relative change in the soil is a concern because of the known toxic effects of Al and its effect on the pH of the soil (Brady 1984).

The second component to be included in a comprehensive nutrient management plan is the proper design and management of manure (and effluent) handling and storage areas. These items have been the subject of extensive research, both in Australia, and overseas (see Bodman 1997; Culley & Phillips 1989; Hu 1995; Norstadt *et al.* 1977, Schulte 1998). This research has led to best management practices for these elements of a feedlot and they will not be considered further in this thesis.

Land application of manure is the third component to be included in a comprehensive nutrient management plan (USDA & US EPA 1998). Experimental data for each unique environment where manure is being applied is required to find the optimum manure application rate, as the ability of the crop to uptake the nutrients in the manure will be site specific (USDA & US EPA 1998). It would be expensive and take considerable time to determine the optimum application rates experimentally (Park & Seaton 1996).

A compartmentalised model incorporating a nutrient balance of inputs and outputs could be used as an alternative, or preferably in parallel with, the experimental approach. Monte Carlo techniques should be used to incorporate the stochastic nature of the variables that drive the system. A model that compartmentalises the nutrients into the various output paths should define the optimal rates of application such that the applied nutrients do not exceed the capacity of the soil and crops.

The time of application is also identified as being important in the land spreading of manure (USDA & US EPA 1998). A Monte Carlo model could be run to observe the effect of applying solid manure at different times of the year.

The land management component of a CNMP requires that proven conservation measures be used where manure is applied to the land. These include soil conservation work, conservation tillage measures, crop residue management, riparian buffer, filter strips, field border contour buffer strips to intercept, store and utilise nutrients that may migrate outside the area of application (USDA & US EPA 1998). The model developed in Chapter 5, assumes that recognised best practices relating to these areas are used.

Record keeping components need to be incorporated into a CNMP to store data on the quantities of manure and effluent produced and applied. Records on soil, crop and water analyses also need to be kept. The use of an electronic database is recommended as the tool to describe and interpret the soil information to aid in the decision making process about the use and management of the land (USDA & US EPA 1998). The environmental monitoring database, described later, is designed to store this information but also to allow easy extraction of relationships that may not be apparent without manipulation of data.

The success of a CNMP depends on the availability of resources to assist in its development and implementation. An environmental monitoring database and simulation model are tools that would be useful in this regard (USDA & US EPA 1998).

2.6 Aspects of Sustainable Design and Management of an Intensive Confined Animal Feeding Operation

Intensively confined animal feeding operations (CAFOs) is a term for feedlots widely used in the literature. In the last 25 years there has been significant research into environmental issues associated with CAFOs, particularly the feeding operations or storage and application of manure and effluent. These studies include:

- detailed investigations into the feedlot pad (Lott & McKay 1990);
- leaching beneath the pad (Sweeten 1991);
- leaching beneath storage ponds and manure stockpiles (Bodman 1997; Culley & Phillips 1989; Schulte 1998);
- the effects of manure on sealing the storage ponds (Sweeten 1991);
- design of the storage systems and sizing the irrigation area (Hu 1995; Norstadt *et al.* 1977);
- runoff from the feedlot pens, manure storage facilities and irrigation areas (Linderman & Ellis 1978; Lott, Loch & Watts 1994);
- dust and odour issues (Azevedo & Stout 1974; Hancock *et al.* 1996; Jones *et al.* 1992 Koelsch 1998);
- manure management (Sander 1996; Schmitt *et al.* 1997; Sri Ranjan *et al.* 1995; Thompson *et al.* 1997);
- manure application (Sander 1996; Sri Ranjan *et al.* 1995; Thompson *et al.* 1997);
- effluent irrigation (Gardner *et al.* 1993; Sri Ranjan *et al.* 1995); and
- tracking the nutrients through the whole system (Eigenberg *et al.* 1995; Gardner *et al.* 1993; Gardner *et al.* 1994).

Despite all these studies, many regulations have been applied to the utilisation of manure and effluent that have not always been based on adequate research (Sander 1995; Thompson *et al.* 1997). Historically, regulations for feedlots have been derived from empirical data originating from the United States, which considered only the environmental system (Lott *et al.* 1996).

Managing intensive CAFOs using a holistic view involves a mass balance and systems approach (Eigenberg *et al.* 1995; Sri Ranjan *et al.* 1995). Some groups are carrying out research on feedlot manure and effluent in a sustainable context (Louisiana State University 1998), but few of these focus on the feedlot system in a holistic context.

Even before animals were confined in feedlots, research had been undertaken to characterise the composition of their manure, its value as a fertiliser and the changes that take place when the manure is stored (Azevedo & Stout 1974 (after Russell & Richards 1917; Sievers 1922)). However, in the early days, most research focused on manure and effluent as a waste disposal problem (Butchbaker n.d.; Gilmour *et al.* 1977; Linderman & Ellis 1978; McKay & Watts 1984; Olsen & Barber 1977; Watts & McKay 1986). Some recent literature also approached the problem from this aspect (Geary & Gardner 1996; Hu 1997).

A conceptual shift by some authors has lead to manure and effluent being seen as a resource to be utilised as a crop fertiliser (Chang *et al.* 1991; Lander *et al.* 1998; Sander 1995; Schmitt *et al.* 1997; USDA & US EPA 1998). The fertiliser value in manure and effluent can be obtained if it is distributed according to crop needs and not lost to the environment (Sander 1995; Thompson *et al.* 1997). These authors define manure and effluent as pollutants only if they are allowed to leave or degrade the system.

A feedlot system has the potential to benefit the production system and the environment. Feedlots allow operators to produce a consistent quality product and they also allow a reduction of land degradation during drought (Zoebl 1996). Cullen (1991a) asserted that any public drought subsidy should be conditional on penning and feeding stock rather than allowing broad acre destruction of the landscape.

Four possible outcomes from the utilisation of resources produced by intensive animal feeding operations are outlined below.

1. The effects will not detract from the value of the local agro-ecosystem or greater environment.
2. The more desirable outcome would be for the agro-ecosystem created by the feedlot to enhance the ecosystem that existed previously, and not detract from the value of the greater environment.
3. A less desirable effect of the feedlot operation would be to enhance the local agro-ecosystem, but degrade the greater environment.
4. The effects of the feedlot cause degradation both to the local agro-ecosystem and the greater environment.

A sustainable system that utilises manure and effluent will have point 1 as its minimum objective. Outcomes 3 and 4 are not considered acceptable in a sustainable framework.

A sustainable intensive animal system as interpreted by Lott *et al.* (1996) is one where nutrients are applied to the soil-plant at such a rate that:

- some build up of nutrient and salt can occur in the soil, however, this build up should not cause the soil to reach toxic levels of any chemical element and the soil remains healthy or has an improved status;
- crops remove a substantial amount of the nutrient and salt; and
- leaching losses are at an acceptable level.

Gardner *et al.* (1994) and Watts *et al.* (1994) used mass balance principles as a means to interpret the nutrient and salt cycling in a feedlot system. These authors adopt the conservation of mass principles and each kilogram of nutrient excreted by the animal is partitioned and explicitly accounted for in the soil, plant, water and air pathways. Gardner *et al.* (1994) identified the pathways of P, N and salts as being the most important for sustainability.

Gardner *et al.* (1994) defined a reuse area that has the ability to adsorb the nutrients and salts for the same period (or longer) as the design life of the feedlot, as sustainable. This definition of sustainability does not consider the stochastic nature of the system and the long-term effects that the operation may have on the environment once the feedlot has ceased operation beyond its "design life". For example, the potential for diminished concentrations of soluble ions caused by inappropriate irrigation practices that leads to leaching. This has an adverse affect on the soil through acidification (Sverdrup *et al.* 1995).

The variability of the system is recognised by Gardner *et al.* (1994) and they suggested that flexibility is required to adjust for errors in the predicted manure and effluent production and application quantities. Suggestions for incorporating this flexibility include; off-site sales of stockpiled manure, increasing the size of the irrigation area or augmenting the effluent irrigation supply with "fresh" water. However, an understanding of the essential characteristics the system and its inherent variability may provide avenues for better management practices that account for the unpredictable nature of the operation.

2.7 Utilisation and Losses of the Nutrients and Salts in Manure and Effluent

Literature pertaining specifically to the use of nutrients and salts contained in manure and effluent from an intensive livestock system is limited for the climatic and soil regimes found in Australia. However, a considerable amount of literature focuses on the use of effluent and sludge from other sources, such as sewage.

A feedlot development is seen to be ecologically sustainable if the system outputs do not pollute water bodies or degrade the soil and ensures land use options for future generations (Lee & Pankhurst 1992; Gardner *et al.* 1994). This is achieved by keeping nutrient losses to an acceptable minimum through utilisation of the nutrients applied. Applicable water quality standards or legislation determines the acceptable levels (Gardner *et al.* 1994).

Water quality standards are contested by Cullen (1991a) who stated that they do not take into account regional variability. Another limitation of water quality standards is that they are often based on concentrations and not loads at the point of discharge, and therefore do no not address the total loads leaving a particular catchment. The criteria used in the standards will result in some areas being in breach most of the time. However, if loose criteria are applied some fragile systems will be destroyed although water quality meets the required criteria. This demonstrates the need for appropriate regional criteria (Cullen 1991a).

The heterogeneous nature of the inputs to a feedlot system makes the prediction of the quality of the output very difficult (Lott *et al.* 1996). This in turn adds to the complexity in developing an understanding of the utilisation of any of these outputs, an example being the salt concentration in the drinking water. Any excess in salt to that required by the animal, passes through the animal to the manure pad and hence through the rest of the system. Drinking water quality will therefore have an effect on the quality of the product that is to be applied to the utilisation area.

Geary and Gardner (1996) contended that mass balances using measured inputs will often provide accurate estimates of the fate of nutrients in a reuse area. Therefore, the mass balance approach predicts the limiting nutrient/salt that can be applied to a land area for a given number of years. However, the mass balance theory does not take into account the variability of outputs and the interactions that may occur between parameters. The build-up of organic matter, which acts as a further sink for the particular nutrient and the increased uptake of the nutrient by the plant, are also not considered in the mass balance approach (Lott *et al.* 1996).

An example of the limited value of this approach is found in the consideration of the addition of salts to the system. If salinity were considered in isolation, then the sustainable life would be based on the point where yield reduction would occur due to the increased concentration of salt in the soil solution. This in itself varies considerably with the salt tolerance of the particular crop that is grown, the design of the irrigation system and the management practices adopted.

Sustainable management of manure and effluent may not necessarily require low rates of application (Lott *et al.* 1996; White & Saffley 1984; Zoebl 1996). The contrary may be required where the aim is to gain maximum economic return through intensive production and hence minimise environmental impact (White & Saffley 1984). When the production system is managed to operate at its optimum performance level, the uptake of nutrients in the crop biomass reduces the potential for nutrients and salts to leave the system via the undesirable runoff and leachate pathways.

Table 2.1 summarises the variables that need to be considered in the sustainable use of the resources produced by a cattle feedlot from the perspective of both the environmental and production systems. Some of the difficulties and attributes gleaned from the literature pertaining to each variable are included and are expanded in Chapter 3.

Table 2.1. The Difficulties and Attributes Associated with Variables to be Considered in use of Manure and Effluent

Variable	Difficulty	Attributes	Reference
Salt	Different concentrations in drinking water - cattle able to drink water that is considerably more saline than that generally considered suitable for irrigation purposes. Different crops are affected at different concentrations before yields are limited. The effect on the soil depends on the soil type.		Gardner *et al.* 1994
Crop Uptake	Different crop types have varying abilities to take up nutrients. Crop uptake is a function of dry matter and nutrient concentration. The nutrient concentration of the plant is also a function of the concentration of nutrients in the soil. Potentially plant uptake can increase to meet application rates within certain limits.		Gardner *et al.* 1994 Lott *et al.* 1996
Nitrogen	Volatile nature makes it difficult to follow the fate of different species of N. This causes difficulty in the mass balance approach when trying to partition the various pathways	• Important nutrient for plant growth	Gardner *et al.* 1994
Phosphorus	Ability of the soil to accumulate P and the effects of increasing organic matter. "Clearly a large organic matter build up acts as a significant store of TP and is a significant store of nutrient." The movement of P in the soil profile is very slow and in some forms is unavailable to the plant.	• Also one of the macro-elements required for plant growth	Lott *et al.* 1996
Organic matter	Nutrients are bound up in the organic matter and mostly unavailable for plant uptake until mineralisation occurs. The rates of mineralisation are dependent on the nutrient and other factors (such as the C:N ratio, temperature and moisture). Total concentration analysis may not take into account that these nutrients are unavailable for leaching or other loss pathways	• Increases infiltration rates and soil water storage capacity. • Increases the cation exchange capacity of the soil • Improved soil structure	Lott *et al.* 1996

Chapter 3. Characteristics of the Agro-ecosystem

In a feedlot, the design of a system to utilise effluent and manure will be the governing factor in laying the foundation for its 'sustainability'. Once the design of the system is satisfactory, the management practices that are adopted become the variables that influence sustainability. In order to define appropriate management practices it is necessary to have a thorough understanding of the nutrient pathways and interactions that occur within the system from the collection, holding and application of the manure and effluent.

The environmental impact on ground water, surface water and soil are the areas of most concern when a feedlot production system is considered and should be central to any definition of best management practices. Parameters that indicate impacts include, nitrate in ground water, salinity of the soil and ground water, and nutrient enrichment of surface waters from excess P and N. Methods of monitoring and interpreting these impacts need to be incorporated into a systems framework that is capable of providing a measure of sustainability.

Characterising the inputs to, and outputs from sub-systems within the feedlot is the first step to defining the sustainability of the utilisation of manure and effluent. The sustainable management of the system then requires that changes of state in the agro-ecosystem be monitored to provide early warning signs of environmental degradation. In this study, the sustainable utilisation of manure and effluent is considered in the context of providing a nutrient and irrigation source to supply the needs of a crop. Other uses of the manure and effluent are not investigated.

3.1 A Systems Framework

Kruseman *et al.* (1996) presented an integrated framework for analysis of sustainable land use at plot, farm and regional level, taking into account the interactions among agro-ecological and socio-economic variables. These authors define a set of hierarchically ordered systems of land use ranging from the plant system through to the cropping system, the farm system, the local system, the regional system to the national or even higher system levels. As a consequence of these levels of systems, different variables should be defined or measured at different spatial and temporal scales when considering the sustainability of a particular system.

A system may be seen to be sustainable at one level but not another. For example, the pollution of ground and surface water at the regional level as a consequence of the over use of fertilisers at the farm level. At the regional level the use of fertiliser is not seen as sustainable, while at the farm level the perception is the reverse (Kruseman *et al.* 1996). In this case, the view of sustainability at the farm level considers only the current production system and not the ability of the system to continue into the future.

The hierarchy of the feedlot system is difficult to describe using the above approach. Feedlots will have a number of cycles that ultimately form part of the hierarchal order. A more appropriate description might consider a number of cyclic systems on several levels. Each of the sub-systems in the cycle have inputs and outputs where some of the outputs may be an input to the next system. The inputs and outputs may also interact with systems at other levels. A schematic representation of these systems is shown in Figure 3.1.

With reference to Figure 3.1, the confined animal system has feed, stock water and rain as inputs and manure and runoff as outputs. Resources of this system are the animals themselves and the facilities (i.e. in the case of a feedlot - the condition of the pen surface, fencing, feed structures and drainage works). The outputs from the animal system become the inputs to the storage system. Within the storage system, natural processes may chemically change the inputs, but in physical terms the outputs from the storage system are the same as the inputs to the storage system.

The next system in the cycle is the cropping system where the inputs from the storage system are utilised to produce feed for the animal system, hence closing the cycle. In all of these systems, inputs and outputs to and from systems at other levels are also occurring. External inputs also occur, with systems having common external inputs that are climatic. Some of the systems have further external inputs such as fertiliser added to the soil system and subsequently the cropping system.

The animal system interacts with the soil and water systems through the movement of its outputs. These outputs contain matter that can be both beneficial and detrimental to the soil system. If there is excessive amounts entering any of the systems there is potential for harm to be done. An understanding of the pathways involved will highlight the areas of significance with regard to potential problems.

Figure 3.2 indicates a lower hierarchal level, showing the interdependent parts of the systems involved in the utilisation of feedlot manure and effluent at the plant-soil level (Lott *et al.* 1996). This figure highlights the many pathways and pools which add to the complexities of the interactions and processes occurring within and between the systems. The following sections look at each of these systems individually.

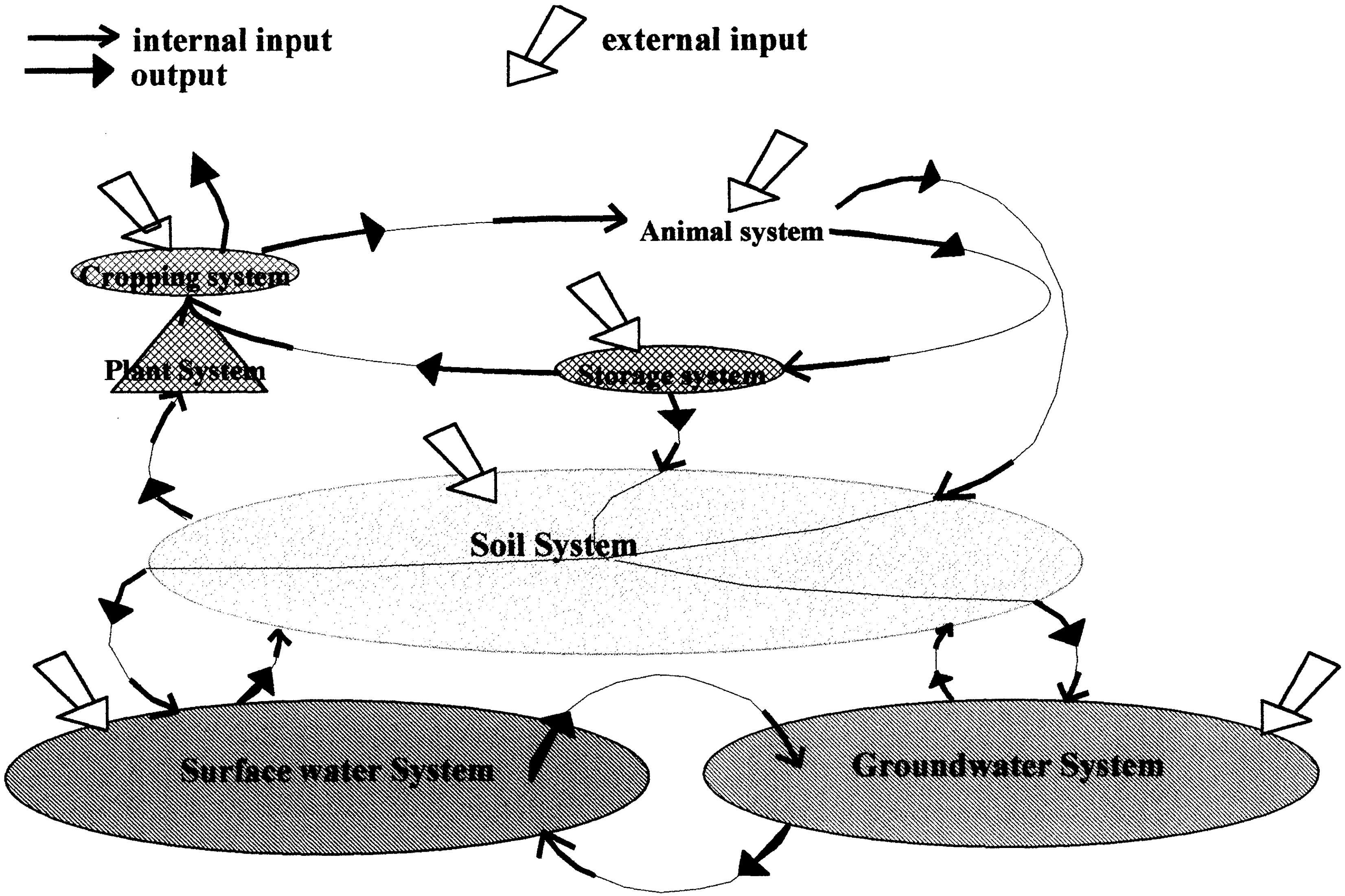

Figure 3.1. Pictorial Representation of Systems Involved in Land Utilisation of Resources Produced by a Feedlot

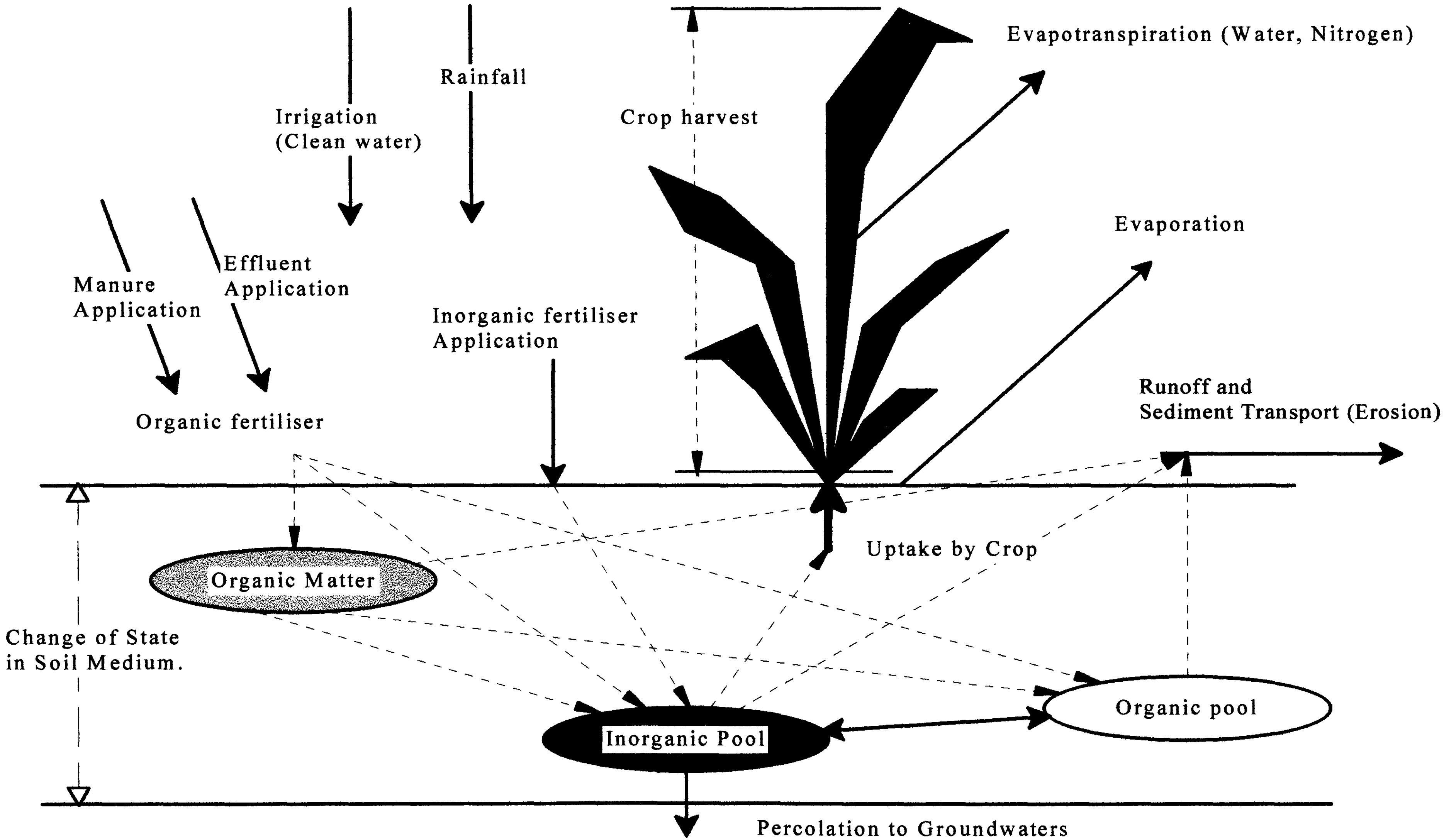

Figure 3.2. The Interactions and Pathways of a Manure/Effluent Utilisation System (Lott *et al.* 1996)

3.1.1 The Soil System

Soil consists of three phases - solid, liquid and gas. The gas and liquid phase make up the pore space and the solid phase occupies the remainder of the soil volume. All three of these phases are important in terms of the transport of water, nutrient and salts throughout the soil profile, beyond the soil profile and from the soil to the plant. The proportion of the volume occupied by each phase effects the structure and hence the behaviour of the soil. As the soil moisture content changes, the relative proportion of the gas and liquid phases also changes (Ellis & Mellor 1995).

Frissel (1978) divided the soil compartment into 3 sub pools, which are outlined below.

1. Available soil pool. This is where plants obtain their nutrients and includes those nutrients in soil solution and on exchange sites. This is the inorganic pool shown in Figure 3.2.
2. The unavailable soil pool. This includes the soil minerals, which are stored in the organic pool shown in Figure 3.2.
3. Residues or soil organic matter pool. Elements in this pool have variable and often long residence times before being mineralised and transferred to the available pool.

While transfer between pools occurs in both directions simultaneously (as is the case for mineralisation and immobilisation in soil) it is usually only possible to measure the net result (Frissel 1978). This transfer of nutrients occurs through the pore space in the soil.

3.1.1.1 The Pore Space

Soil pores consist of both the liquid and gas phase and the volume occupied can be changed through pressures being applied to the soil (e.g. compaction). Pore space is an important component in the transfer of nutrients, either to plant roots or ground water and an understanding of the essential characteristics of soil pore space is required.

The structural characteristics of soil are a product of the formation of the aggregates or peds and the forces that bind these aggregates together. For the transport of nutrients, the pores between the aggregates are of greater interest than the peds themselves. The extent to which gases and the soil solution are mobile in the soil is dependent on the range of pore sizes, their continuity and their tortuosity (Greenland & Hayes 1981).

Greenland and Hayes (1981) divided the pore space into three groups, depending on the equivalent diameter of the pore and hence function of that pore size. The smallest pores have an equivalent diameter of less than 500 nm and can retain water against both drainage and the suction exerted by plant roots. The larger storage pores have an equivalent diameter in the range 500 nm to 50 μm and retain water against gravity, but allow it to move to roots. Transmission pores, which are the largest, have an equivalent diameter greater than 50 μm and are unable to retain water against gravity, which allows soils to drain after rain or flooding. At other times these pores are air filled.

The movement of gases and liquids is a function of the pore sizes. The gaseous phase (also termed the soil atmosphere or the soil air) consists of a mixture of gases derived from the aboveground atmosphere and from the respiration of soil organisms and plant roots. Microbial and root respiration is able to take place in the soil due to the movement of gases through the transmission pores.

The liquid component is known as the soil water solution and is the transport mechanism for material in both suspended and dissolved form. The most direct influence on the transport of nutrient from the soil solution to the plant comes from the storage pores. Nutrients that are associated with inorganic and organic colloids as exchangeable ions are released to the soil solution when the organic matter is mineralised. These and other nutrients in solution in the soil water are immediately available to plants from these pores. The nutrient ions move into plant roots partly by mass flow of the solution to the root, and partly by diffusion (Greenland & Hayes 1981).

3.1.1.2 The Soil Zones

In order to discuss the transport of water, nutrients and salts through the soil profile, several zones can be identified by the effect they have on the flux of the components through the system. Ellis and Mellor (1995) defined four major zones in the soil through which the water will move. They are the soil zone, which lies nearest the surface and controls the infiltration of water into the ground. Beneath this is the intermediate zone in which percolation of water is the dominant process. The thickness of this zone varies and is dependent on relief and rock type. This overlies the capillary fringe, in which most of the pores are filled with water. These three zones comprise the vadose zone, which is the most important in terms of the processes occurring in the soil. For example, the cation exchange process and consequently the amount of each cation in solution and thus available for leaching.

The saturation zone lies beneath the vadose zone and the water table denotes the interface between this and the capillary fringe. The root zone, which is of most interest in terms of plant growth, generally occurs in the first two zones. If the saturation zone impinges on the root zone, problems of aeration will occur, with the soil becoming anaerobic.

3.1.2 The Crop System

The four biological cornerstones of a sustainable cropping system are (Roberts 1995):

1. the reduction of soil loss;
2. the maintenance of organic matter;
3. the avoidance of nutrient loss; and
4. the minimisation of toxic accumulation.

With the addition of manure and effluent it is imperative to define the best management practices that are able to balance 3 and 4.

The plant or crop system interacts with the soil system through the root zone of the soil. The solid phase of the soil provides the structural stability for the plant, while the water phase provides the turgidity and nutrients necessary for plant growth. The gaseous phase provides oxygen required by the plant for the photosynthesis process, which transports carbon out of the soil. Plants obtain many of their nutrients through the exchange of cations, with the nutrient cations taken up through the root system in exchange for hydrogen ions (Ellis & Mellor 1995). The cation exchange capacity of, and cation balance in the soil is important because of the potential adverse effects that an undesirable balance or limited exchange capacity can have on the structural properties of the soil. These effects will be discussed in Section 3.2.9.

Nutrient cycling within a plant system varies according to the plant type, and in most intensively grazed or cropped system nutrients spend only a small portion of the overall cycle time in the plant compartment. On the other hand, in the plant matter of indigenous vegetation or forests, a portion of the nutrients may remain in vegetative form for long periods of time. It therefore becomes difficult to define the size of the nutrient pool in these systems (Frissel 1978).

The concentration of nutrients in a particular plant depends on the species of plant, climate, soil type and management practices. Most major nutrients typically fall in a relatively narrow range in the harvested tissue of most crops (Kardos *et al.* 1977). Typical N concentrations range from 1.2% to 2.3%, P concentrations range from 0.3% to 0.5% and K ranges from 0.2% to 2.4% for most crops (Kardos *et al.* 1977). Reuter and Robinson (1986) presented the ranges of nutrients found in different crops, for particular plant parts. Some of these ranges are larger than those presented by Kardos *et al.* (1977), though they are within the same order of magnitude.

Organic matter is composed of residual plant matter or animal manure and it comprises primarily carbon and nutrients that can be a reserve of nutrients required by crops. Continuous input of organic matter enhances the nutritional status of the soil, particularly with respect to N (Leeper & Uren 1993; Syers & Craswell 1995). The amounts of nutrient supplied to the crop depend on the type of manure, how it is handled, the crop grown and the growing conditions. Nutrients not released in the first growing season will be slowly available to crops over the following seasons.

Plants build up protein from simple substances, which are used in a variety of essential processes necessary for plant growth such as carbohydrate utilisation, stimulating root growth and uptake of other nutrients. All plants except for the *leguminosae* and a few other minor groups obtain their N requirements from the ammonium cation (NH_4^+) or the nitrate anion (NO_3^-). *Leguminosae* plants, in symbiosis with nodule bacteria, obtain most of their requirements from the gaseous N circulating through the soil (Leeper & Uren 1993; Wild 1988).

3.1.3 The Ground Water System

The current knowledge of the basic processes of an ecosystem is considered by some authors to be fundamentally distorted by the perceptions formed from the visible elements and an ignorance of the unseen elements, for instance, the ground water (Lee & Pankhurst 1992). A better understanding of the processes involved in impacting the environment, particularly the environment that is not visible, is a prerequisite of managing a sustainable system.

Ground water can exist at varying depths below the surface. It can be confined and is therefore pressurised or unconfined and able to rise unimpeded to the surface if sufficiently recharged. Water may also move up through the capillary zone to the plant roots from a ground water table that is close enough to the root zone. Ground water may also flow laterally where it can migrate to surface water storage. The movement of water in the water table is saturated flow and is therefore governed by the saturated hydraulic conductivity of the soil and the hydraulic pressure gradient. In the vadose zone, flow is sometimes saturated, but mostly unsaturated and its rate of flow is dependent on the matric potential of the soil solution.

Lateral movement of ground water may move nutrients and salts from one area to another and the nutrient that is of major concern is N, specifically the nitrate ions. Nitrate moves freely in ground water and contamination from nitrate is long lasting, as remediation is difficult and expensive. Adult humans are usually only affected if nitrate is reduced to nitrite, however high concentrations of nitrate in drinking water (~10 mg/L nitrate-N) can lead to methemoglobinemia in infants up to 3 months old (Pierotti 1996). Animals can also suffer from a similar disease if there is an excessive concentration of nitrate in their drinking water. However, animals are able to tolerate up to ten times as much nitrate as humans, with concentrations of up to 100 mg/L nitrate-N being acceptable for a livestock water source (Pierotti 1996). Because of the immobility of the P ion excessive concentrations of this nutrient are generally not considered as being a problem nutrient in ground water.

3.1.4 The Surface Water System

The surface water system consists of man made storages, natural lakes, creeks, streams and rivers. These water bodies are replenished by runoff from the catchment, migration from ground water, or transport via mediums such as pipes or drains. This transport can be gravity driven or a hydraulic head may be imparted to the water by means of a pump. Water may also infiltrate from the sides and bottom of the storage and become ground water.

The quality of the surface water will be a function of the route it takes from when it lands on the soil, as rainfall or as applied irrigation, to the time it reaches the storage. If the majority of the travel time is spent as overland flow, the opportunity time for mineral-water contact will be somewhat less then if the majority of the travel time is through the soil profile. The constituents of the soil at the depth through which the water flows will also govern the amount of dissolved ions present in the water (Trudgill 1977). However, if the velocity of water following an overland route is intense enough to cause erosion of sediments from the soil, elements that are strongly absorbed to the soil particles such as P, are likely to be transported to the surface water storage.

Water chemistry is unbalanced by the entry of P and N rich runoff, which can result in eutrophication (Emsley & Hall 1976). Eutrophication literally means good feeding and in a water environment this can be ecologically disastrous as the species being well-fed increase in population to such an extent that they deplete the dissolved oxygen in the water, causing other species to die. The system becomes further unbalanced when more oxygen is consumed in coping with the decay of dead organisms and excreta of the over-populated species. P and N are the limiting growth factors in many aquatic systems and their abundance in the runoff entering the water body will be the cause of eutrophication (Emsley & Hall 1976). An aquatic ecosystem may handle the stress imposed upon it by the entry of polluted runoff for some time without any obvious or undesirable symptoms. This cumulative stress may result in a sudden collapse of the ecosystem and the emergence of a new and less desirable ecosystem (Cullen 1991b).

3.2 Processes that Affect the Transfer of Nutrients and Salts in the Agro-ecosystem

Agro-ecosystems are immature ecosystems maintained at early successional stages in order to exploit their high production to biomass ratio and the accumulation of living material (Edwards *et al.* 1993). In a mature natural ecosystem the nutrient cycle is a relatively closed system because of the efficiency with which these ecosystems can trap and hold nutrients. An immature ecosystem does not hold nutrients tightly and the nutrients have a tendency to escape from the system (Edwards *et al.* 1993). The nutrient retention rate of the ecosystem is a function of many factors, including the soil type and drainage characteristics, irrigation and management practices and cropping regime. On a typical livestock farm, the proper use of manure may result in about 75% of the nutrients in the harvest being recycled (Bauder & Jacobsen 1990).

The transfers of nutrients between the systems that encompass the agro-ecosystem occur as a result of conditions, processes and controls that are internal and external to the systems. These controls can be physical, chemical and/or biological. As energy regulates photosynthesis and temperature, the chemical and biological processes in soils and plants that are temperature dependent are governed by the energy flow. As a result of the energy flow, the systems are dynamic and hence continually changing their biological, physical and chemical environments. Internal control is exerted as the component parts of the system respond to this shifting equilibrium (Frissel 1978).

The nutrient cycle, which consists of plant uptake, growth, consumption, decomposition and release, is dependent on biological factors for its completion (Frissel 1978) and the movement of water controls the rate at which the majority of physical, chemical and biological changes occur (Ellis & Mellor 1995). Therefore, the process that drives or effects all the other process and interactions is the hydrological cycle, as it deals with the cycling of water through the systems.

3.2.1 Hydrological Cycle

Water is important as a transport medium because of the interactions between nutrient solubility and the supply of water, which in turn controls the nutrient availability for both plants and micro-organisms (Frissel 1978). All output pathways that have the potential to be harmful to the environment have water as the transport vector.

The hydrological cycle begins with precipitation, which passes along a number of possible pathways and is eventually returned to the atmosphere via evaporation or transpiration. This continuous cycle involves water in its three phases - solid, liquid and gas. It reaches the earth as a liquid (raindrops) or solid (snow or hail) and returns to the atmosphere as a vapour. The possible pathways in between the start and the end of the cycle are listed below.

1. Interception by an obstruction above the earth's surface such as leaves. This water will be stored temporarily on the surface of the obstruction and eventually fall to earth or evaporate back to the atmosphere, thereby short-circuiting the 'in between' pathways of the hydrological cycle.
2. Overland flow: Water flows over the land ending up in a surface water storage, such as a lake, dam, river or creek and eventually the ocean. From the surface water storage it may be evaporated back to the atmosphere or migrate to the ground water. Infiltration into the soil may occur as water flows over the land. This pathway is of most importance in the erosion process.
3. Infiltration into the soil: The top layer of soil governs the infiltration rate of water into the soil. From within the top layer the water can move along a number of paths. The cycle may be completed by evaporation directly from the soil or water can be taken up by the plant and returned to the atmosphere in the transpiration process. Water may flow vertically up or down as a result of the water potential or it may move laterally through any of the soil zones to end up as surface water from where it can be evaporated back to the atmosphere. If the soil has a dispersive nature, this pathway may also be important in the erosion process, especially if tunnelling develops.

Figure 3.3 shows some of the possible pathways for the movement of water after it falls as precipitation. Pathways 2 and 3, described above, are of the most interest in terms of nutrient and salt transfer through the systems. The residence time of the soil solution in the root zone is important for the cycling of nutrients and in particular the pathway from the soil solution to the plant. The travel time for water in the pathways discussed above and shown in Figure 3.3 maybe short (hours) or very long (years) (Frissel 1978).

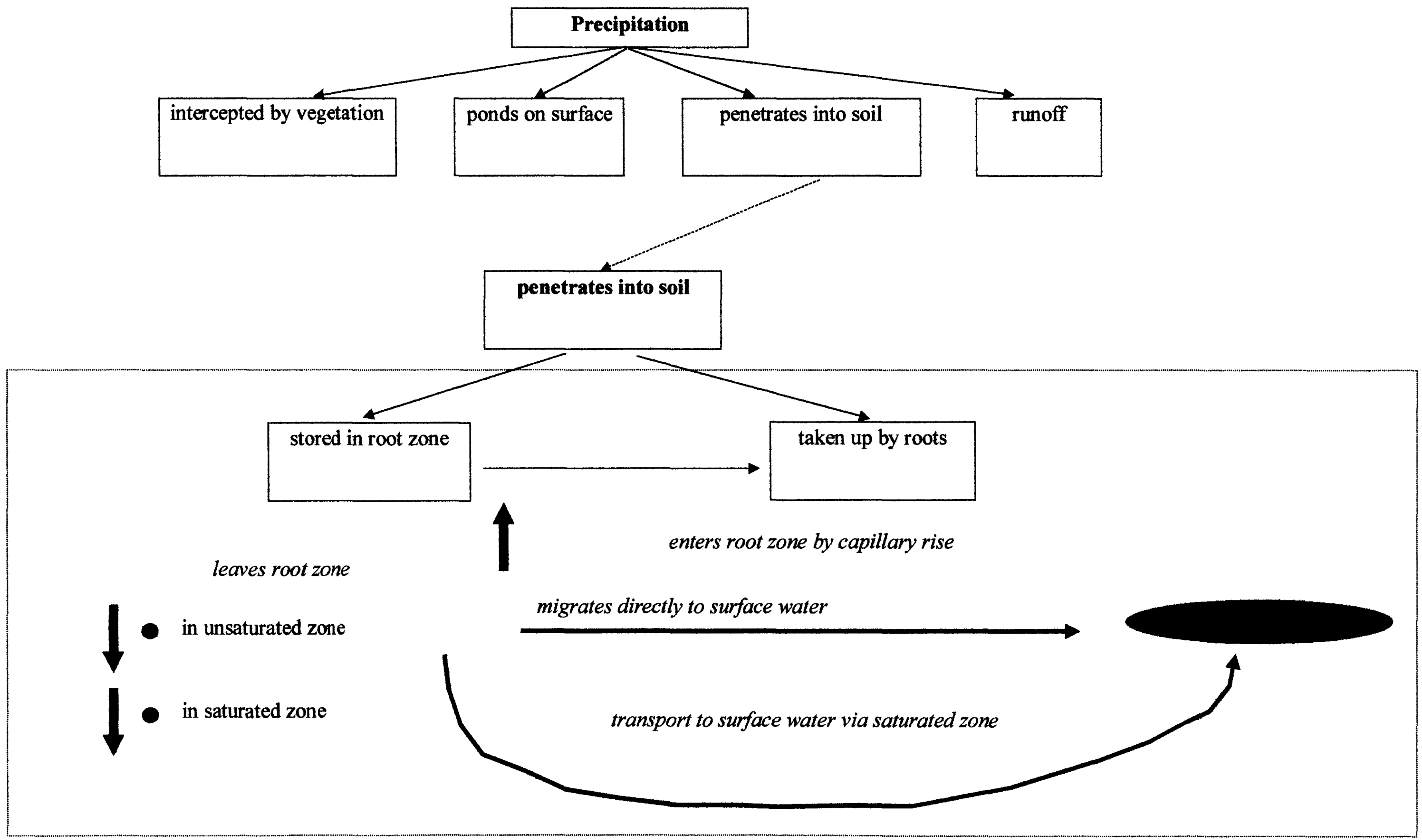

Figure 3.3. Some of the Pathways in the Hydrological Cycle

There are four forces that control the movement of water into, through and out of soils - gravity, adsorption, capillarity and osmosis. Matric suction is the combined effects of adsorptive and capillary forces (Ellis & Mellor 1995). The capillary (from the Latin word for hair) force can work in all three dimensions (downwards, upwards or sideways) and water travels through very fine spaces as a result of this force (Leeper & Uren 1993). The smaller the pore, the more tightly it holds water.

The quantity of water passing through the soil can be described by Equation 3.1 (Wild 1988):

$$Q = W_p - W_r - W_{ev} - W_{tr} - \Delta W \qquad \textbf{Equation 3.1}$$

where

Q is the amount of percolation,

W_p is rainfall plus irrigation,

W_r is runoff,

W_{ev} is evaporation from soil,

W_{tr} is transpiration from the crop, and

ΔW is the change in water content of the soil.

Losses from the soil system through erosion, overland spills and leaching are driven by flow of water through the system. The use of the simple water mass balance in Equation 3.1 (Wild 1988) in a model is one option of quantifying these losses. However, there are other dynamics and processes, also affected by the flow of water, which influence the cycling of nutrients through the systems under consideration.

3.2.2 The Nutrient Cycle

If the transfer of a nutrient into and out of the system is in balance, then the system is in a steady state for that nutrient. If nutrient transfers are not equal then the system is accumulating or declining in respect of that nutrient. The system may not necessarily be in the same state of balance for all nutrients (Frissel 1978). To quantify the nutrient cycle it is necessary to define the physical and chemical properties for each element and the quantity and rate of transfer of the nutrient along the pathways that cross the boundaries or are within each system (Frissel 1978).

Edwards *et al.* (1993) stressed the importance of recognising the strong link between the availability of organic matter and both bio-diversity and nutrient cycling. The systems of interest in terms of the cycling of nutrients in manure and effluent are the soil, crop, ground water and surface water systems. Water movement is the vector that interconnects these systems and controls the rates of change that occur in the manure and effluent (Trudgill 1977). Manure and effluent contain significant amounts of organic matter and it is the changes that occur when this organic matter is decomposed that are of importance to the systems and the cycles within the systems.

3.2.3 The Organic Component

The reserves of soil organic matter should be maintained at least at the levels that are the minimum necessary to protect the soil and maintain productivity. The sustainability of agricultural production systems depends on this maintenance and requires knowledge of the characteristics of soil organic matter (Syers & Craswell 1995). This knowledge is essential for understanding the structure, fertility and chemical reactions that occur in the soil and the impact from the addition of manure (Whitbread 1995).

Organic matter derived directly from plant residue generally contains 25% dry matter. Of this approximately 44% is carbon, 40% oxygen, 8% hydrogen and 8% ash or inorganic elements (including N). The above constituents of organic matter form various compounds and the approximate breakdown of these compounds is (Brady 1984):

- carbohydrates - 60%;
- protein - 10%;
- fats, waxes and tannins - 5%; and
- the remainder lignins.

The crude proteins contain approximately 16% N and small amounts of S, Fe, and P (Watts *et al.* 1994).

Manure applied to the soil as a fertiliser is a source of soil organic matter and contains nutrients such as N, K and P. These nutrients are an important source for plants and the breakdown of the organic matter also releases cations such as Ca, Mg and K. As well, organic matter commonly accounts for about half of the cation exchange capacity of surface soils (Skjemstad *et al.* 1987; Wild 1988) and has the ability to reduce exchangeable Al (Syers & Craswell 1995), and therefore reduce Al toxicity. The characteristics of inorganic fertiliser additions to a system are that of a pulse, however the addition of organic matter as the nutrient source provides a more steady supply (Roberts 1995).

The factors that affect the amount of nutrients and organic matter contained in manure that has been stockpiled or composted, when it is applied to the land area are (Lott, Powell & Sweeten 1994):

- the salt content of livestock drinking water and rations;
- the length of time the manure has been stockpiled or composted;
- the amount of air excluded by compaction (stockpiled);
- the moisture content of the stockpile and friability of the manure;
- the amount of aeration or oxygenation of the manure (compost);
- the prevailing weather conditions when being spread (loss to the atmosphere of manure dust in adverse conditions); and
- when the manure is incorporated into the soil after application.

Organic matter is the site of biological activity and influences the physical and chemical properties of the soil out of proportion to the small percentage usually present (Syers & Craswell 1995). The maintenance of the levels of the organic matter in the soil is dependent on inputs and maintaining optimum levels of soil organic matter should be the aim of sustainable management practices (Syers & Craswell 1995). This requires an understanding of the different forms of organic matter and the processes that affect these forms.

Agricultural soils generally have lower levels of organic matter than similar soils under native vegetation (Syers & Craswell 1995). This is due to a lower input of plant carbon through residues (i.e. litter fall), tillage and other agricultural practices that increase the rate of decomposition through mixing the surface soil and increasing the number and intensity of the wetting and drying cycles (Neeteson & Van Veen 1987; Roberts 1995; Syers & Craswell 1995). A study in Asia indicated a period of 20-30 years of bush fallow was required to restore the 20-30% of soil organic matter lost during a 2-year cropping period (Syers & Craswell 1995). The breakdown of additions of manure would accelerate the replacement process to some degree.

Ellis and Mellor (1995) quoted 5% of the total volume occupied by the soil as a typical amount of organic matter present in topsoil. This relates to the case where organic matter is derived from the breakdown of plant material and soil organism residues. Leeper and Uren (1993) gave some representative values for Victorian soils ranging from 0.6 % total organic matter for a gradational calcareous of the Mallee region to 10.5 % for a gradational non-calcareous soil in the Western District of Victoria. Leeper and Uren (1993) also showed the variation of organic matter with depth of soil with a krasnozem varying from approximately 6.6% in the top 25 cm to approximately 1% at a depth of 80 cm. Typical agricultural soils in Australia have organic matter contents in the range of 0.5 – 2.5% (Lott *et al.* 1997b).

Organic matter can improve the soil physical properties and hence increase the stability of soil (Skjemstad *et al.* 1987; Wild 1988), increase infiltration, reduce runoff and decrease erosion (Syers & Craswell 1995). It also acts as an energy and nutrient source for decomposer organisms in the form of soil fauna and microflora (Skjemstad *et al.* 1987; Wild 1988). This has important positive effects on the physical and biological characteristics of the system (Syers & Craswell 1995), as the decomposer organisms break down the organic matter in a form that the plant can readily take up.

The decomposer organisms are bacteria and fungi that reduce organic matter to humus through a series of reactions such as mineralisation. This process is known as humification. The materials that remain from the process are highly modified parts of plant tissues and bio-synthesised complex organic molecules or humic materials that are resistant to further breakdown. These materials vary greatly in their chemical nature and molecular weight and a proportion of them combine with the mineral fraction of the soil to form organo-mineral complexes of varying complexity and stability. It is these complexes that are largely responsible for the increased short-term stability of some organic fractions (Skjemstad *et al.* 1987).

Once organic material has been added to a soil, it will undergo decomposition unless this is prevented or the material is already fully decomposed (Ellis & Mellor 1995). The rate of decomposition is a function of the level of the substrate to feed the soil biomass and environmental conditions, such as length of growing season and climate, especially temperature and rainfall (Leeper & Uren 1993; Neeteson & Van Veen 1987; Syers & Craswell 1995). Therefore, decomposition of organic matter is dependent on complex interactions of the physical, chemical and biological processes within soils (Whitbread 1995). However, cultivation can cause a rapid decline in the organic matter (Neeteson & Van Veen 1987) due to the continual removal of plant matter and exposure of physically protected organic matter to decomposition (Leeper & Uren 1993).

Rose (1991) reported that the greatest effect on the physical structure of the soil from adding organic matter in the form of manure is to decrease the bulk density and increase the water holding capacity of the soil. However, there are conflicting reports on the effects of the availability of water in a soil after the addition of manure. Some studies report significant increases and some suggest that the amount of available water is not greatly changed (Mbagwu & Piccolo 1990; Rose 1991).

Rose (1991) used natural aggregates packed in columns, to investigate in the laboratory the effect of long term applications of farmyard manure on the physical properties of soils (the actual length of time manure was applied is not specified). Six pairs of soils were used from United Kingdom field experiments at Rothamsted, Saxmundham, Wellesbourn and Woburn. One of each of the soil pair had a long record of heavy farmyard manure application. Soil samples were taken from the top 15 cm of the profile in late summer. The results obtained by Rose (1991) showed that the organic carbon content of the soil increased with long-term applications of farmyard manure. The soils used in the experiment were a clay loam, sandy clay loam, sandy loam and a silty clay loam. The texture of the soil from Saxmundham changed from a clay loam to a sandy clay loam. The increase in organic carbon for each soil type is given in Table 3.1.

Table 3.1. Increase in Organic Carbon in the Soil (Investigated by Rose (1991))

Soil Type (without manure)	Soil Type (with manure)	Ratio increase in organic carbon
Sandy loam	sandy loam	1.66
Sandy loam	sandy loam	2.08
Clay loam	sandy clay loam	2.03
Clay loam	clay loam	3.59
Silty clay loam	silty clay loam	3.23
Silty clay loam	silty clay loam	3.45

Other parameters measured by Rose (1991) were bulk density, soil density, proportion of solids, total porosity, macro-porosity, micro-porosity, aggregate porosity, field capacity and wilting point. For each soil, the effect of the addition of farmyard manure was in the same direction. The biggest effect was the increase in total porosity due to the greatly increased micro-porosity, though there was a decrease in macro-porosity. Porosity of the aggregates and the proportion of total pore space within the aggregates were increased and results showed the addition of farmyard manure has beneficial effects on the physical properties of the soil (Rose 1991).

3.2.4 Carbon

All organic matter has carbon as one of its constituents. The carbon cycle begins when CO_2 is assimilated by higher plants through the photosynthesis process. The plants convert CO_2 into numerous organic compounds, such as carbohydrates, lignins, protein, fats etc. These organic compounds eventually reach the soil as plant residues, in the waste products of animals and humans or as their remains. In the soil the micro-organisms and soil animals digest the organic matter which releases nutrients for plants. Humus and CO_2 are the relatively stable end products of this process. The other sources of CO_2 in the soil are from the respiration of rapidly growing plant roots, and rain water. Under optimum conditions more than 100kg/ha of CO_2 may be generated in the soil per day. The more common production rate is in the range of 25-30 kg/ha per day (Brady 1984).

A small fraction of the CO_2 undergoes chemical reactions to produce carbonic acid (H_2CO_3) and the carbonates and bicarbonates of Ca, K, Mg and other bases. These bicarbonates are readily soluble and may be removed in drainage or used by the plant. However, the greater part of the CO_2 in the soil eventually escapes to the atmosphere and completes the cycle when it is again assimilated by plants (Brady 1984).

3.2.5 Nitrogen

More than 90 to 95 % of the total N in topsoils occurs in organic compounds. The remaining N is present as nitrate and ammonium ions (Wild 1988). Organic N exists in a variety of materials, chemical forms and locations in the soil that differ in their susceptibility to decomposition and release of mineral N (Skjemstad *et al.* 1987). Wild (1988) reported on the average percentage of chemically combined N in the top 15 cm of soil profiles. For temperate regions the figure is around 0.1 to 0.3 %, and for arid regions generally less than 0.1 % of chemically combined N is in the top 15 cm of soil.

The deposition of N in rain and from the combustion of fossil fuels usually amounts to less than 5 kg N/ha/year (Wild 1988). N gains in soil also occur as a result of adsorption of ammonia from the atmosphere, with 4 kg N/ha/year being measured at Rothamsted, England (Wild 1988). Values are likely to be greater where atmospheric concentrations are higher - near feedlots, grazed pastures and industrial centres (Wild 1988).

Nitrogen is an essential component of proteins and related amino acids. It is critical as building blocks for plant tissue and in the cell nuclei protoplasm in which hereditary control is created (Brady 1984). Plants and many micro-organisms in the soil synthesise amino acids from simple organic compounds. Proteins are constructed when amino acids join together to create chains through the formation of peptide bonds. Peptide bonds are formed through the reaction of adjacent carboxyl (-COOH) and amino ($-NH_2$) groups with the elimination of water. These groups attach to the α-carbon and hydrogen or another organic group, represented by R in the structure of the amino acids shown in Figure 3.4 (Zumdahl 1990).

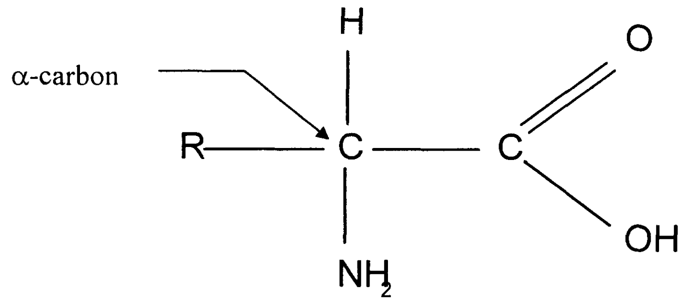

Figure 3.4. Structure of Amino Acids - Building Blocks of all Proteins (Zumdhal 1990).

When organic matter is added to the soil it contains organic N in protein compounds. These nutrient proteins are broken down into their constituent amino acids, which are then used to construct those products that organisms need as a source of energy. Ammonia (NH_3) is produced through this process which is known as enzymatic hydrolysis. Heterotrophic micro-organisms are responsible for enzymatic hydrolysis. The reaction that takes place is shown in Equation 3.2, where the R consists of the carboxyl group plus hydrogen, methane or more complex substitutes (Zumdahl 1990). Further reactions then occur with ammonium and carbonate ions being the end products. The ammonium ions are held in the soil solution and on clay and organic matter exchange sites. This and the intermediate reaction is shown in Equation 3.3 (Brady 1984; Wild 1988; Zumdahl 1990).

$$R\text{-}NH_2 + HOH \xrightarrow[\text{hydrolysis}]{\text{enzymatic}} R\text{-}OH + NH_3 + \text{Energy} \qquad \text{.......... Equation 3.2}$$

$\uparrow$ Amino Combination

$$2NH_3 + H_2CO_3 \longrightarrow (NH_4)_2CO_3 \rightleftharpoons 2NH_4 + CO_3 \qquad \text{.......... Equation 3.3}$$

The conditions favoured for this series of reactions, known as ammonification, are found in warm, well-drained (moist but not wet), aerated soil, which have plenty of basic cations present. However, it will take place to some extent under almost any conditions, because of the great number of different organisms capable of accomplishing such a change (Brady 1994). Immobilisation is the reverse of this process and after stubble incorporation there may be net immobilisation for 3 weeks to 3 months (Hoare 1992). The mineralisation rate is a measurement of gross ammonification minus immobilisation (Boyle & Paul 1989).

Ammonia (NH_3) does not normally appear in the soil in the free state, but as the ammonium cation (NH_4^+) which, when it attaches to the colloidal material, becomes only slowly available to plants (Leeper & Uren 1993; Wild 1988). Both organic and inorganic fractions have the ability to fix ammonium in forms that are relatively unavailable to plants or micro-organisms. In the 2:1 clay silicate mineral most cations are able to move freely into and out of the crystal where they are attracted due to the internal negative charge (i.e. they are exchangeable). However, the ammonium (and potassium) ions fit into the cavities that exist between the crystal units and they become fixed as a rigid part of the crystal. This prevents the normal expansion of the crystal and the ions are held in a non-exchangeable form and are only slowly released to plants and micro-organisms. With the clay content generally being higher in the subsoil than the surface soil for any given soil type, ammonium fixation is greater with increasing depth in the profile. This may be considered a means of conserving soil N and so is an advantage in situations where the supply of N is limited (Brady 1984).

Nitrification is the next series of reactions that take place. They occur when the NH_4^+ supply exceeds the needs of the heterotrophic organisms that are present in the soil. NH_4^+ is taken up readily by plants if it is not absorbed by the colloid, used by the heterotrophic organisms or subjected to nitrification. Nitrification by autotrophic bacteria will occur if aeration is good and is the process of enzymatic oxidation of ammonium ions to nitrates.

Certain micro-organisms in the soil bring about the reactions involved in the nitrification process in two coordinated steps. The first step is the production of nitrite ions mainly by autotrophic bacteria (e.g. *Nitrosomonas spp.*) which is generally immediately followed by the second step; the oxidation of the nitrite ions to the nitrate form by another group of autotrophic bacteria (known as *nitrobactor*). Hydrogen ions are produced by the first step resulting in an increase in soil acidity (Brady 1984). Generally the second step follows so closely to the first that there is no accumulation of the nitrite ions, which is fortunate as large concentrations of this ion is toxic to higher plants. However, a problem can develop in very alkaline soil or where large ammonia fertiliser applications have been made. In this situation the levels of ammonia may be toxic to the Nitrobactor bacteria (but not the nitrite organism) and will therefore not be available to act as the catalyst for the second step in the process. This will result in nitrite accumulation of sufficient magnitude to have an adverse effect on plant growth or to encourage gaseous losses of N (Brady 1984). The two steps involved in the nitrification process are shown in Equation 3.4 and Equation 3.5 (Brady 1984).

$$2NH_4^+ + 3O_2 \xrightarrow[\text{oxidation}]{\text{enzymatic}} 2NO_2^- + 2H_2O + 4H^+ + \text{Energy}$$ .... **Equation 3.4**

$$2NO_2^- + O_2 \xrightarrow[\text{oxidation}]{\text{enzymatic}} 2NO_3^- + \text{Energy}$$ .... **Equation 3.5**

When adequate ammonium ions are available and conditions are ideal (temperature in the range of 27° - 32° C, and suitable moisture content), nitrification can occur at a very rapid rate and the supply of nitrate more than meets the needs of a crop. When excess nitrate is produced it is lost in drainage or runoff, either in a soluble form in the soil solution, or by volatilisation. Nitrate can also be reduced to nitrous oxide or dinitrogen gas through denitrification, which is mostly a biological reduction (Wild 1988).

Volatilisation is the gaseous loss of N from soils as dinitrogen, nitrous oxide and ammonia. The flux of water-soluble gases of nitrous oxide and ammonia from the soil-plant system to the atmosphere depends on the partial pressure of the particular gas in the atmosphere and its concentration in the soil solution. Volatilisation is governed by the rate at which the gas is transported away from the surface and is therefore determined by wind speed and evaporation. The flux is reversible, depending on the concentration of ammonia in the atmosphere. The ammonia dissolves in the soil water to form the ammonium ion.

In areas where there are distinct wet and dry seasons, ammonification of organic compounds occurs, but the soil is generally too dry for nitrification and ammonium accumulates in the soil. In the early part of the wet season the conditions are conducive to rapid oxidation due to the alternate wetting and drying of the soil and there is flush of nitrate (Wild 1988), which becomes susceptible to leaching into the ground water if there is sufficient rainfall. To take advantage of this initial flush, crops have to be planted at the right time. In the context of utilising manure and irrigating with effluent, this could have important consequences if the planting window is not accurate in terms of the ability of the crop to uptake the nitrate and prevent its accession into ground water.

Ammonia levels are an important regulator of nitrification as high levels constrain the second step of the process. If the carbon:nitrogen ratio of the residue added to the soil is high, this will prevent the release of ammonia and therefore nitrification does not occur. The process of nitrification is also retarded by both very high and very low moisture contents. The optimum moisture content for plant growth is generally the optimum moisture content for nitrification (Brady 1984). However, the factor that controls the transfer of N in the soil and its availability to plants is the proportion present with respect to the amount of carbon present - the carbon:nitrogen ratio.

3.2.6 Carbon:Nitrogen Ratio

The organic carbon to organic nitrogen ratio, the C:N ratio, expressed as a ratio by weight, is relatively constant for different soils under a wide range of management conditions (Wild 1988). Excluding soils that are strongly acid ($pH<5$), and poorly drained soils, the C:N ratio of topsoils falls within narrow limits mostly between 10:1 and 14:1 (Wild 1988). Mature plant material has a higher ratio and C:N ratios greater than 14:1 are a strong indicator of partially decomposed plant material in the soil (Brady 1984; Leeper & Uren 1993; Wild 1988).

The C:N ratio generally decreases down the profile and values of approximately 5:1 are not uncommon at depths of one metre or so (Wild 1988). This may be due to the presence of fixed NH_4^+ as the organic carbon content of most soils decreases with depth but the content of fixed NH_4^+ remains constant or increases, so that the C:N ratio narrows (Brady 1984; Wild 1988).

The C:N ratio has a selective influence on the micro-organisms in the soil. The C:N ratio of the soil is increased by the addition of carbohydrates, which increases the microbial oxidation of the carbon compounds. As the population of the micro-organisms increases, they utilise the inorganic N present more rapidly than the plants, hence immobilising the inorganic N. No ammonium nitrogen is released and nitrification is at a standstill. Serious competition with plants is initiated and mineral N concentrations may decrease (Brady 1984; Leeper & Uren 1993; Wild 1988). As the carbonaceous matter decomposes the available food supply reduces and microbial activity slows down, reducing the C:N ratio. Some of the immobilised N will then be mineralised and ammonium compounds will again appear in the soil (Brady 1984; Wild 1988). Lance (1986) showed that the carbon to nitrate-N ratio determines the extent of the denitrification process.

The amount of nitrification occurring in the soil when organic matter in the form of manure is incorporated is strongly dependent on the C:N ratio. Nitrification regulates the amount of nitrate that is available for leaching to the ground water. There is usually a temporary loss of mineral N if the incorporated material contains less than about 1.8 % N, which corresponds to a high C:N ratio of about 30:1 (Wild 1988). Gardner *et al.* (1994) quoted a C:N ratio of approximately 25:1 for manure, but the age and origin of the manure is not identified.

Immobilisation will occur when organic matter with a high C:N ratio is added because there is sufficient carbon present to stimulate microbial growth and all the N added in the organic matter is incorporated into the microbial cell structure (Gardner *et al.* 1994). In the ensuing weeks microbial respiration reaches a peak and results in the loss of carbon as carbon dioxide and the C:N ratio again reduces (Wild 1988).

Understanding the N cycle is important for the timing of manure additions and planting of crops. To obtain the maximum benefit from the N that is contained in the manure it is also important to have a knowledge of its C:N ratio and the effects this has on the immobilisation of the N. Timing of manure additions should be such that the crop utilises all the N available and a minimal amount is available to be leached out of the profile.

3.2.7 Phosphorous

Phosphorous and hydrogen (as water) are commonly the limiting elements in Australian agriculture (Leeper & Uren 1993). Humans tend to upset the natural rhythm of the P cycle (organic and inorganic) by discharging P wastes into water instead of returning them to the land (Emsley & Hall 1976). P then becomes a pollutant in water where it has a destructive effect on the ecosystems of lakes and rivers.

The species best able to use phosphate in surface water is algae and if there is a sufficient supply of N an increase in phosphate will result in an algal bloom. These blooms can occur when certain natural processes suddenly increase phosphate concentrations but the algal blooms quickly reduce the lake's natural phosphate concentration and growth ceases before any real damage is done. However, if the supply of phosphate is continuous, a high algal population can be maintained and the water becomes eutrophic. The bloom then blocks out the sunlight from deeper plant life preventing photosynthesis which would replenish the deeper layer with oxygen. At the same time the dead algae use up what oxygen remains (Emsley & Hall 1976).

Phosphorus is an immobile element, which readily binds to the soil particles and its susceptibility to leaching is considered minimal. The tendency for P to bind to the soil particles has important implications for the management of the environment, particular with regard to erosion control measures.

Hayes *et al.* (1990) reported an increase in P concentration in soil irrigated with secondary-treated-municipal-sewage effluent, while the P content of the same soil under the same conditions decreased when irrigated with potable water. High levels of P can lead to the production of insoluble compounds of Ca, Mg, Fe and Zn. These elements are less available to plants in these insoluble forms (Hayes *et al.* 1990). This has important implications in terms of managing the cropping system.

3.2.8 Salts

Leeper and Uren (1993) stated that the history of failed irrigation schemes in Australia as a result of salinity is because of an inadequate provision for drainage. This causes a rise in the water table and hence through capillarity, salt water reaches the surface and evaporates, producing salt in the surface soil. However, an example is given of one early irrigation scheme on the Werribee River, which is virtually free of salt and water logging problems. The success of this scheme is attributed to the good drainage provided by the deep seams of gravel underlying the area (Leeper & Uren 1993). This 'natural' drainage system was able to transport the leached fraction to the nearby sea.

Hayes *et al.* (1990) compared the effects of irrigation with effluent to that with a potable supply on turf grass grown in a gravelly sand loam in Tuscon, Arizona. These researchers found that the Na levels in a soil irrigated with effluent were significantly higher than the soil irrigated with a potable water supply. These high salt concentrations can be toxic to plants, and when the concentration of salt is greater in the soil solution then in the plant, the effect is to draw water from the plant to try and equilibrate the two concentrations. This is commonly referred to as the osmotic potential of the soil solution.

The predominate salt of concern in agriculture in southern Australia is sodium chloride which commonly makes up two thirds or more of the total soluble solid of a soil (Leeper & Uren 1993). Saline soils also have a presence of bicarbonate and sulphate and sodium carbonate is another salt that can be important. The effects of sodium carbonate may be due to its concentration but its more likely effect is due to the consequences of the high pH that it brings about (Wild 1988). The soil structure also tends to become water-unstable, which adversely affects permeability (Leeper & Uren 1993).

3.2.9 Cation Exchange

Colloids, both organic and inorganic are the sites for cation exchange. The colloids have a large specific surface area (large surface area relative to mass) and are able to form stable suspensions in water. As the particle size decreases, the specific surface area increases and the proportion of potentially reactive atoms in the solid increases. Therefore, the likelihood is greater for the solid to participate in chemical reactions.

The ease with which a cation is adsorbed depends on the number of valent electrons that it possesses and the degree of hydration of the ion. Cations with a high valency have a high energy of adsorption and are adsorbed in preference to lower valency cations. When there are two cations of equal valency the radius of hydration will govern which cation is adsorbed. The cation with the smallest radius of hydration is adsorbed preferentially as it can get closer to the particle surface as shown in Figure 3.5 (Ellis & Mellor 1995).

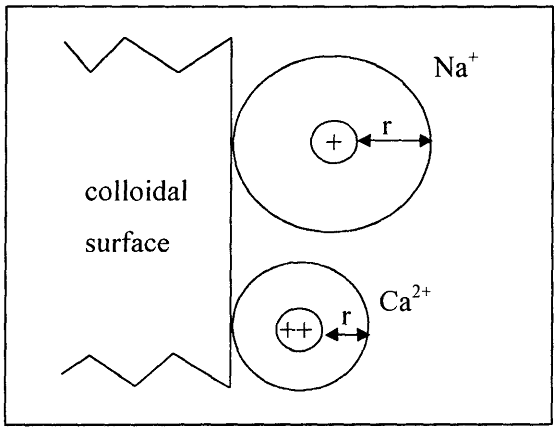

Figure 3.5. Factors Influencing Cation Adsorption Preference; r = Radius of Hydration (Ellis & Mellor 1995)

The generally accepted sequence of preferential adsorption for base cations is:
$Ca^{2+} > Mg^{2+} > K^+ > Na^+$ (Ellis & Mellor 1995).

As well as serving as sites for cation exchange, organic colloids aid in the release of nutrients such as Ca, K and Mg from soil minerals. This occurs when the humus colloid is saturated with H^+ ions and is therefore acidic. This acid is comparatively strong and removes the base nutrient from the molecular structure of the mineral. This base nutrient is then loosely adsorbed onto the organic colloid and becomes more easily available to higher plants (Brady 1984).

Leeper and Uren (1993) reported on an experiment that looked at the effect of urine on the soil pH and the concentrations of various cations and anions in the soil solution, four hours and 64 days after the addition of urine. In the first four hours there was a rapid increase in pH (from 5.8 to 7.3) which is said to be due to the rapid hydrolysis of urea. The concentration of all the ions normally found in urine (Ca^{2+},Mg^{2+}, K^{+}, Na^+, NH_4^+, Cl^-, SO_4^{2-}, NO_3^-) also increased markedly in the first four hours. However, the nitrification process reduced the concentration of NH_4^+ after 64 days (approximately 1/5 of the four hour level), while the concentration of NO_3^- increased nearly 10 fold after 64 days from the four hour level. The concentration of Ca^{2+} also increased ten fold and there were also increases in the Mg^{2+} and Na^+ levels. These increases of the exchangeable ions are attributed to their replacement with the H^+ on the exchangeable sites. The Ca^{2+}, Mg^{2+} and Na^+ cations are then available to be leached from the system or utilised by the plant.

3.2.10 Soil Erosion as a Vector of Nutrient and Salt Transfer

Erosion of soil causes a loss of nutrients and salts from the soil system. Cultivation generally leaves the soil bare of a vegetative cover before there is a crop canopy and after harvest. During this time it may be subject to water or wind erosion.

Erosion is a function of rainfall (amount and intensity), runoff, wind, soil, slope, plant cover and the presence or absence of conservation measures. All these factors combined provide a multitude of possible outcomes from any given set of parameters and in the study of erosion it is difficult to isolate any cause and effect principles that would simplify the understanding of the process (Morgan 1995).

Three phases of soil erosion can be identified. The first is the detachment of individual particles from the soil mass and the second is their transport by the erosive agents of running water or wind. Deposition is the third phase if sufficient energy is no longer available to transport the particles.

Rainsplash is one of the most important detaching agents with weathering, mechanical and biochemical processes also contributing to the detachment of the individual particles from the soil mass. Some of these include alternate wetting and drying, freezing and thawing, frost action, running water and wind. The soil is loosened by all these processes making it more easily removed by the transport agents (Morgan 1995).

The pathway of the water through the vegetation cover and subsequently over the land has a direct effect on the erosion process and therefore the hydrological cycle controls the processes of water erosion. Direct throughfall is the term given to water that lands directly on the surface as a result of there being no vegetation cover or because it passes through the gaps in the vegetation. Water that is intercepted by the canopy can be returned to the atmosphere by evaporation or it can reach the ground via a number of pathways. Leaf drainage is the term given to the water when it reaches the ground by dripping from the leaves and stemflow refers to the water running down the stem to the ground. Direct throughfall and leaf drainage both produce rainsplash, however vegetation cover dissipates the energy of raindrops and therefore throughfall has a greater energy potential to produce rainsplash.

3.2.11 Nutrient Pathways in Land Areas Receiving Manure and Effluent

The best agricultural soils have organic binding substances that are produced during the biological decomposition of organic residues in the soil (Martin & Focht 1977). Organic matter is also a source of nutrients. When the plant does not take up these nutrients they have the potential to be lost to the system through the pathways of leaching and runoff.

Hyland (1995) emphasised that nutrient depletion in Australian soils, which are naturally low in most nutrients in the first place, is an important form of land degradation. Hyland (1995) estimated that in 1992/93, 1.2 million tonnes of nutrient were removed from the soil in agricultural products as N, P or K. Only 0.98 million tonnes were returned as fertiliser, a 26 % shortfall in one year. This imbalance is not likely to be uniform on a regional basis as grazing areas and some cropping zones are continually exporting the nutrients with minimal replacements. There are likely to be excess nutrients in areas where there are intensive agricultural industries.

Kardos *et al.* (1977) stated that the accumulation of waste product constituents in a productive agricultural soil should not be detrimental to the ability of the soil to produce food and fibre. The same authors and others (Chang *et al.* 1991; Olsen & Barber 1977) report on studies that found depressed grain sorghum yields when high rates of animal manure are applied. The Kardos *et al.* (1977) study was on a clay loam soil in West Texas treated with 269 and 538 Mg/ha of animal manure. Salt accumulation in the soil as a result of the application of manure was pinpointed as the cause of the decreased yields. Soil salinity and soil structure deterioration have also been identified as being caused by the application of feedlot runoff (Swanson *et al.* 1973).

Chang *et al.* (1991) studied the effects of different rates of manure application under irrigation and compared them to the effects of no irrigation. The manure was added to a clay loam soil at Lethbridge, Alberta, Canada for eleven years. The rates used were equal to one, two and three times the recommended rates for the soil type, under non-irrigated (30, 60, and 90 t/ha) and irrigated (60, 120, and 180 t/ha) conditions. The rates are on a wet weight basis, with a very high average moisture content of 82.5% ± 21.8% (SE). Total irrigation applied in a year was 20 cm or less in 5 cm increments as required. Both regimes had control plots of zero application of manure. The effects of different tillage regimes were also examined. All the parameters measured either increased or remained static with the exception of pH which decreased. The effects on the parameters generally were noticed to a greater depth under irrigation. Chloride and nitrate effects extended to 150 cm in both the irrigated and non-irrigated cases. The tillage regimes generally had no effect on the parameters monitored. Chang *et al.* (1991) concluded that the maximum recommended application rates of manure for both irrigated and non-irrigated agriculture in the soil type and climate studied is inadvisable because of the potential pollution problems associated with the accumulation of N, P, Cl, Na, Ca, Mg and Zn in the soil profile.

Rose (1991) in the experiment of paired soil samples described previously, found variable effects on the pH after applying manure. For the two pairs of silty clay loam soils, one increased in pH and one decreased following the addition of manure. The pH of the clay loam pairs both decreased, however there was an increase in pH for both the silty loam soil pairs. This could be attributed to different leaching characteristics in the soils. Nitrification of ammonium and the production of organic acid during decomposition of the organic fraction of the manure were attributed to the decrease in soil pH observed by Chang *et al.* (1991).

Chang *et al.* (1991) found the total and available P increased greatly in the top 30 cm and warn that this concentration of high available P might interfere with uptake of micro-element nutrients by the crop. A much greater increase in total and available P was also evident in the top 30 cm of the irrigated compared with the non-irrigated plot (Chang *et al.* 1991).

Chang *et al.*(1991) also found an increase in the sodium adsorption ratio (SAR) of the soil with increasing manure applications, however different patterns with depth were observed for the different application rates and the irrigated and non-irrigated plots. Na^+, Ca^{2+} and Mg^{2+} generally increased with increasing application rates, except around the 120 cm depth for both irrigation and non-irrigation where the effects of application rates were less than at the other depths. The increased SAR of the soil was coupled with a decreased pH and was attributed to the precipitation of Ca^{2+} as sulfates, phosphates, and carbonates and the displacement of $Ca^{2+} + Mg^{2+}$ on the exchange complex by Na^+ (Chang *et al.* 1991).

The amount of elements present in manure and effluent is dependent on the design of the drainage and storage system and the subsequent management practices that are adopted (Kardos *et al.* 1977). Composting and storage of the manure can reduce the C:N ratio considerably (Kardos *et al.* 1977), which has implications for the mineralisation of the organic N when this composted material is applied to the utilisation area. The reduced C:N will result in more of the N being liberated in the early stages of decay, as the supply of carbon as an energy source for the organisms responsible for the breakdown of the organic matter will be reduced and their numbers will not be subjected to the same population growth that would be seen with a material of a higher C:N ratio. Less N will be incorporated by the organisms and more will be mineralised and available for plant uptake and leaching.

Mobile soil water that is charged with anions creates a favourable leaching environment (Trudgill 1977). For example, the nitrate ion can react with solid phase calcium carbonate in calcareous soil and Trudgill (1977) quoted studies that have demonstrated cation transport through soils can be largely regulated by the production of mobile anions, especially bicarbonate. The cations combine with the anions in solution and are then transported through the soil profile.

3.2.11.1 Crop Removal of Nutrients

The removal of an element from the utilisation area by the harvesting of a crop is dependent on the uptake of that element by the plant and on the amount of plant that is removed in the harvesting procedure. Kardos *et al.* (1977) quoted studies that show corn grain removes between 1 to 3 kg/ha/year of chloride, while the same crop harvested for silage removed 35 to 40 kg/ha/year

Olsen and Barber (1977) reported on a study that applied up to 700 Mg/ha of manure, with crop yields declining when rates exceeded 204 Mg/ha. Grain sorghum yields were reduced when under irrigation and the rate exceeded 134 Mg/ha, with 22 Mg/ha being adequate for maximum yield. Nitrate pollution hazards were found to be minimal only when the crop utilised most of the applied N.

Kardos *et al.* (1977) quoted studies that recorded a 17% recovery of N by barley grown in a soil treated with fresh manure, compared to a 40% recovery from soil treated with fermented manure. Comparisons of the residual effects from the solid and liquid fractions were also noted in the same study. For the liquid fraction the N was immediately available and was largely recovered in the first crop. The N tended to become available over a much longer period of time for the solid fraction (Kardos *et al.* 1977). This is due to the C:N ratio and the dynamics of the organic matter breakdown over time. Knowledge of the composition of the manure and effluent that is to be applied to a utilisation area is therefore very important from the perspective of understanding and predicting the nutrients pathways and their rates of movement through the system.

Crop removal rate can also be effected by excessive rates of effluent application. Sweeten (1991) reported on a study that determined the uptake of N as 74% of applied, P as 41 % of applied and K as 74 % of applied when swine lagoon effluent was applied at recommend soil N needs. This decreased to 33, 17, and 32 % of each of the nutrients respectively when the application was four times the soil/plant requirements (Sweeten 1991). The changes of state in the soil were not reported in this study.

Increases in soil salt concentrations, deterioration of soil structure, reduced soil aeration and water infiltration can result from the over application of manure or effluent. The follow on effects from this are a reduction in crop yields and consequently a reduced crop removal rate. Crop lodging and the possibility for excessive concentrations of a nutrient in a crop to cause it to be unpalatable or toxic to livestock are other effects (Bauder & Jacobsen 1990).

3.2.11.2 Runoff from Land Utilisation Areas

The quality of an aquatic ecosystem is dependent on its catchment, with critical determinants of water quality being land use decisions and management practices adopted (Cullen 1991a). Runoff quality from an irrigation area is dependent on the quality of the irrigation source, salinity status of the soil, climatic variables, water demand of the growing crop and the nutrients that are transported (Thomas & Law 1977). Nutrients and pollutants become dissolved in the runoff (Sweeten *et al.* 1991) and the quality of runoff water is characterised by the concentration of nutrients, particularly P and N, which effect the quality of the receiving water (Thomas & Law 1977).

Cullen (1991b) reported on some typical estimates of non-point P from several types of catchments. These are given in Table 3.2 below and assume a high standard of land management. Examples of inadequate management standards are indicated by a lack of erosion control measures in agricultural catchments and sediment traps in urban catchments. In situations where land management is not of a high standard the estimates should be increased five to ten times.

Table 3.2. Typical Estimates of Non-Point Phosphorus (Cullen 1991b)

Catchment	kg P/ha/yr
Forests	<0.1
Poor quality grazing	0.2
Intensive grazing	0.6
Urban Areas	1.2

Rainfall events that produce high runoff flows generally transport high nutrient loads. Results from an experiment that looked at the influence of flow on the exports of P to Monkey Creek (Sydney) show the importance of these high flow events. 61% of the P and 41% of the water, which moves down Monkey Creek, moves in 1% of the time (Cullen 1991a).

Constructing structures to intercept, reroute and/or store upslope runoff can reduce polluted runoff volume. Soil conservation has been shown to significantly reduce off site contamination from surface runoff and sediment losses. In a study on a Mexico soil with 3% slope, it was found that no-till management reduced sediment N losses by 10-fold, compared to a conventional till corn system (Bauder & Jacobsen 1990).
Skilton *et al.* (1998) found that runoff from a sewage effluent land disposal system operating in the Brisbane bayside suburb of Cleveland, contained an average of 2.4 mg P/L in runoff. The effluent was applied at the rate of 3000 mm/yr and contained an average of 10.8 mg P/L and therefore the land disposal scheme was removing approximately 75% of the applied P. The effluent was applied to pasture, which was slashed but not removed from the site. This immobilised the P that was taken up by the plant and after 15 years a peat layer had developed on the surface. The results of this experiment led the authors to question the use of P sorption curves as a method of designing the life of a sewage effluent land disposal scheme. This would seem a valid conclusion considering unaccounted losses of P (~3.5 t/ha) over the course of the operation of the scheme (>20 yrs). However, the concept of land disposal is very different to one of crop utilisation, which can be seen from the very high rates of effluent application.

Other authors (Edraki *et al.* 1998) measured low N concentrations (5 kg/ha) in runoff from a plot experiment at the same location (Cleveland, Brisbane), in which eucalyptus and pasture were irrigated with sewage effluent. As a percentage, water losses from runoff were nearly double the losses from drainage or interflow, with approximately 20% of that applied being lost through runoff. The mechanism for generating runoff was found to be a function of saturation excess, rather than infiltration excess. Knowledge of the infiltration characteristics of the soil is an important component in a sustainable manure and effluent utilisation scheme.

3.2.11.3 Nutrient and Salt Transfer Resulting from Infiltration

The gravity and capillary forces that draw water into the soil govern the infiltration rate. Sands and sandy loams generally have higher infiltration rates than clay soil because of the larger spaces between the soil particles. However, large cracks in clays can cause them to have infiltration rates higher than would be expected (Morgan 1995). If the soil is dry the infiltration rate is high but as the pores in the soil become filled with water the capillary forces decrease. The infiltration rate then reduces to a level that represents the infiltration capacity. The infiltration capacity is the maximum sustained rate at which water can pass through the soil. In theory this corresponds to the saturated hydraulic conductivity of the soil, however air entrapped in the soil pores as the wetting front passes downwards through the soil tends to make the infiltration capacity lower than the saturated hydraulic conductivity (Morgan 1995).

Frissel (1978) highlighted the varying travel and residence times that water can encounter in soils. The migration of water through the root zone to the water table and then to surface water may cover several kilometres, whereas the pathway of infiltration, plant uptake and transpiration may be only a few centimetres long. To highlight the extremely long travel times that may occur the following example was given by Frissel (1978)-: Annual rainfall of 25 cm is transported to a creek via the exchange of ground water and surface water over a distance of 300 m. For a water-filled pore volume of 33% the mean water velocity is approximately equal to 25/0.3 = 75 cm/year. Therefore, the residence time of the water in this pathway is 300/0.75 = 400 years. This suggests that present day contamination may still be making its way to our drinking water in hundreds of years time and highlights the importance of understanding the behaviour of the soluble nutrient component in the soil.

Flow rates of water, and therefore nutrients, through the soil are heterogenous and can be complicated by the presence of stagnant or dead water in the soil (Frissel 1978). Minerals that result from weathering or applied in irrigation water are concentrated by evaporation and become locked up in the stagnant phase. These minerals can only reach the moving-water phase if the water content in the soil becomes so high that the stagnant phase disappears. Iwata *et al.* 88) in looking at the application of Darcy's law to unsaturated flow stated that when unsaturated flow is observed microscopically, the parts where flow exists are saturated, but there is no moving water in the other parts where flows do not exist.

The total dissolved salt in effluent and manure added to the soil is of concern because of the effects that salt has on plant growth and water systems. Irrigation management requires that a leaching fraction be included so that the amount of salt entering the root zone is equivalent to the amount of salt leaving. This will prevent a build up of salt to levels that become toxic to plants (Gardner *et al.* 1993). However, in the absence of a system to collect the drainage water, leached salts are likely to end up in the ground water.

Manure applications can provide plant nutrients and improve soil physical and chemical characteristics, but increase the potential for ground water contamination (Bauder & Jacobsen 1990). The manure also has a greater nutrient holding potential because of its increased cation exchange capacity. However, the application of excessive amounts of manure or effluent can result in water with excess nitrate-N concentrations percolating through the soil profile to the ground water. Nitrate is the only form of N that is leached out of the soil in appreciable amounts. This is because ammonium ions are held on clay and humus colloids and are only displaced when cation exchange takes place as a result of the application of a salt solution, whereas nitrate is very soluble in the soil solution and not adsorbed by soils (Wild 1988).

The balance of nutrients in the added manure and effluent will not necessarily be the balance required by the plant, and there may be excesses of some and deficiencies of other nutrients. This requires that the utilisation of a resource, such as manure and effluent, be managed in a manner that will lead to improvement in soil fertility and greater production capability, without causing detriment to other resources. Monitoring the soil, ground water and surrounding surface waters will provide indicators of impending problems associated with the utilisation of the manure and effluent.

3.3 Managing the Utilisation of Effluent and Manure

Soil erosion is a major land degradation issue in Australia, causing a loss of soil from the landscape, a reduction in soil structure and the removal of nutrients and salts. The eroded soil becomes a problem downstream, where the sediment reduces the capacity of rivers and drainage ditches, increases the risk of flooding and reduces the design life of reservoirs. Nutrients attached to the eroded soil particles become pollutants.

To reduce the impact of erosion on the downstream environment requires soil conservation measures, particularly to reduce the transportation and hence loss of phosphorus that is bound to the eroded soil particles. This can be achieved to some degree by crops fully utilising the nutrients and salts provided to maximise production. With beef cattle feedlots, reduction in downstream impacts can be achieved through effective design and management of the manure and effluent utilisation area.

The environmental impacts from soil erosion can be reduced through sound conservation practices applied to areas where manure and effluent are applied. These include terraces, strip cropping, and conservation tillage. Conservation practices such as field borders, grassed waterways, sediment basins, and vegetative filters help to minimise the transport of nutrient and organic material off-site especially if there is a runoff event immediately after the application of manure. However, this situation may be unavoidable at some point, despite the best management practices. Therefore, the design and management of the irrigation system with respect to minimising the losses in runoff is imperative (Barker & Zublena 1993).

The fundamental concept should be one of a management practice focused on utilising the resource rather than simply a mechanism for disposing of a problem. Bauder and Jacobsen (1990) contended that applying manure to crops to simply get rid of it, is not sound manure management. Barker (1993) asserted that carefully managed application of livestock manure slurries on pasture and crop land can effectively utilise plant nutrients in manure without threatening the environment. However, if crops are not regularly harvested and removed from the site, nutrients will recycle back into the soil system and eventually become pollutants (Barker & Zublena 1993).

Attention needs to be directed to the nutrient imbalances in beef cattle feedlot manure to maximise crop growth. The plant N:K:P requirements of a oats or sorghum crop is in the order of 10:8:1 (Reuter &

Robinson 1986), whereas the relative concentrations of these elements in feedlot manure are extremely variable with an average of approximately 7:1:5. Feedlot runoff is also different having an N:K:P ratio of 1:4:10 (USDA SCS 1992). If manure and effluent are added in quantities that meet the nutrient needs of the plant there are bound to be excesses of some nutrients that can't be utilised and a potential exists for pollution. An application rate based on the most limiting nutrient may require the addition of organic fertiliser to meet the crop needs.

To achieve best management practices requires feedlot operators have an area of available land that is sufficient to achieve nutrient balance (Sweeten *et al.* 1991). The balance should be such that there is not an excess of any nutrients that cannot be utilised by a crop, with N and P being the major nutrients requiring consideration. Achieving a nutrient balance requires knowledge of crop uptake rates and soil nutrient status to determine appropriate application rates for a land area. Crop irrigation and application schedules should be site specific and be related to the specific crops and soils. General farming operations and the weather must also be considered in the timing of land spreading with runoff being controlled (Bauder & Jacobsen 1990).

Further best management practices may include the maintenance of the irrigation, drainage and runoff holding systems as necessary to ensure holding pond volume capacities are not decreased and effluent and manure that is held can be used in a timely manner. Feedlots and the utilisation area should be located an appropriate distance from waterways, with grass filter strips along the waterway used to remove nutrients. This needs to be considered in the design phase of the feedlot. A buffer area around bores and ground water recharge areas is also needed and Sweeten *et al.* (1991) suggested that at least 50 metres distance is required between lagoons and ponds.

One of the management practices proposed by Barker and Zublena (1993) was to map the soil type and determine the infiltration characteristics for each field that is to receive manure and/or effluent. The maximum irrigation rate or the amount of effluent that can be applied before runoff occurs will depend on these characteristics. An expensive option for determining the optimum rate of manure or effluent application is to set up an experiment for each identified soil type. Alternatively a simulation model can also be used to investigate the important processes.

The above discussion highlights the importance of understanding the interactions and processes that can occur in the system (Lee & Pankhurst 1992). The need for continuous monitoring is important to provide flexibility in managing potential hazards in a time frame that will circumvent the problem.

3.4 Monitoring Requirements

Monitoring is required to measure those parameters that indicate impacts the system has on other systems and the environment. The measuring of inputs and outputs is important to facilitate the understanding of processes occurring in the system, which would not otherwise be apparent. For a monitoring program to meet the criteria of indicating the sustainability, or otherwise of a system, the indicators used should (Hamblin 1995; Lott *et al.* 1997a; Park & Seaton 1996):

- characterise environmental conditions;
- describe processes or hazards;
- diagnose impacts and reactions to stresses;
- be key variables which are relevant and measurable in time and space;
- be readily accepted; and
- be simply defined, balanced and reported regularly in order to produce meaningful time series.

Monitoring nutrient uptake by plants allows the adequacy of current agronomic practices to be assessed (Lisle & Blair 1996). Further Lott *et al.* (1996) suggested that the nutrient status of the plant can be used as an indicator of soil nutrient status. The rationale is that a plant with a high concentration of a particular element indicates a luxurious supply available in the soil and therefore the application of manure and effluent should be managed such that these levels are controlled.

Using plant nutrient status as an indicator of soil nutrient status requires caution as plant uptake of a particular nutrient can be influenced by the presence of other nutrients in the rhizosphere. There may be an abundant supply of a particular element but the presence of another element may prevent its uptake. Conversely the presence of one element may increase the uptake of another element, with the concentration in the plant tissue being a disproportionate representation of the amount which is present in the soil. Some examples of antagonistic interactions, are K and Mg, K and Ca, Ca and Mg, Fe and Mn, Cu and Fe, P and Zn, P and Fe. Examples of synergistic variations are P and Mo, NO_3 and Ca (Lisle & Blair 1996).

Monitoring the nutrients in runoff requires data on water flow and nutrient concentration. Flow rate has a greater impact than concentration in determining the exports and therefore should be measured more frequently than concentration (Cullen 1991b). However, natural seasonal and annual variations in the system need to be considered. P concentrations in the Thredbo River (NSW) have been measured in the range from 4 to 40 μg/L (Cullen 1991a). These values were taken upstream of the ski villages where there is no obvious human impact. Load is the volume of water multiplied by the concentration and at low flows the load is going to be minimal regardless of the concentration, therefore concentration measurements should be flow based, rather than time based (Cullen 1991b).

In order to determine the safe and optimum application rates of manure and effluent, an integral part of the management is the estimation of the nutrient content and nutrient availability in the manure or effluent (Barker 1993). Manure and effluent vary in their nutrient, salt and water content. These and other quality factors are governed by the age of the animals being confined, the feed ration, manure handling and climatic factors (Bauder & Jacobsen 1990). Barker and Zublena (1993) suggested that the manure and effluent that is to be utilised should be analysed twice annually for nutrient and mineral content. Accurate application rates, in terms of meeting crop requirements can be determined from a laboratory analysis on the manure and effluent to take into account the amount of available nutrients.

The dry matter of stockpile manure is approximately 25% (wet basis) (Watts *et al.* 1994). Of this dry matter, approximately 70 % is volatile solids and 30 % is ash. Watts *et al.* (1994) reported approximately 1.5 % dry matter for liquid effluent, with volatile solids and ash being present in approximately equal amounts. The composition of manure with respect to the macro and microelements is given by several authors (Chang *et al.*1991; Spallacci & Boschi 1984; Watts *et al.* 1994; Westerman & Overcash 1980). Chapter 6 reports on the composition of manure and effluent at the Tullimba feedlot and Chapter 4 will use data collected over the period of this study to investigate the processes and interactions highlighted in this chapter.