

**KNOWLEDGE REVIEW OF MODELLING
NON-POINT SOURCE POLLUTION IN
AGRICULTURE FROM FIELD TO
CATCHMENT SCALE**

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Water Research Commission



Knowledge Review of Modelling Non-point Source Pollution in Agriculture from Field to Catchment Scale

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The first of two report to the Water Research Commission on the project
"Modelling non-point source pollution in agriculture from field to catchment scale:
A scoping study"

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EXTENDED EXECUTIVE SUMMARY

1. INTRODUCTION TO THE PROJECT

1.1 BACKGROUND AND MOTIVATION FOR THIS PROJECT

It is increasingly recognised that non-point source (NPS), or diffuse, pollution plays a major role in the degradation of water quality; specifically with respect to salinity, eutrophication (nutrient enrichment), sediments, pathogens, pesticides and some heavy metals. It is furthermore, increasingly accepted that it is infeasible to properly manage water quality without addressing the contribution from non-point-sources. Consequently, attention is increasingly devoted to the quantification of NPS pollution and to identify means to control it cost-effectively at source.

Agriculture has both locally and internationally been implicated as a major source of NPS pollution because most of the land area is utilised for agricultural activities. It is therefore necessary to assess the contribution that the different agricultural activities make to the different manifestations of NPS pollution, to devise the means through which these can be controlled and to determine and predict the effect that control measures will have to reduce NPS pollution. Understanding the production, delivery, transport and use components of agriculture-derived NPS loadings of water resources and being able to predict the fate of agriculture-related NPS constituents are discrete research themes in the agricultural water domain.

The contribution that irrigation activities make towards the salinisation of water resources has been studied for quite some time and is currently still receiving attention. Other NPS water quality issues of concern that are potentially aggravated by agricultural activities are eutrophication (through fertiliser leaching and wash-off from human settlements), sediments (as a result of erosion), pathogens (from intensive animal production units), pesticides (through the application of insecticides, fungicides and herbicides) and some heavy metals. The present level of knowledge concerning these NPS water quality issues is not as advanced as for irrigation-induced salinisation. In order to plan future research activities and, specifically, a project that will address those issues that require priority attention, it is necessary to take stock of the present situation with regard to NPS pollution and the existing knowledge base as far as agriculture's contributions are concerned. This document describes such a scoping exercise and knowledge review, executed with the involvement of primary researchers, academics and professional practitioners in these fields in South Africa.

1.2 OBJECTIVES OF THE PROJECT

General objective of the project:

- To involve major players active in this field in a scoping exercise and knowledge gap analysis in order to develop the terms of reference for a longer term project that would establish an integrated model for the prediction of agricultural non-point source (NPS) pollution from field to catchment scales for the major agricultural NPS pollutants.

Specific objectives of the project:

- Produce first order estimates of the contribution that agricultural activities are making towards NPS pollution.

- Assess the level of understanding concerning the processes operating at various scales from the microscopic level up to catchment scale, for the main sources of agricultural NPS pollution.
- Identify the linkages between pollutants and processes that can mitigate or exacerbate the impacts.
- Assess the range of mitigating measures that are available to address each of the major sources of agricultural NPS pollution, and
- Assess the predictive (modelling) ability that is available to address each of the major sources of agricultural NPS pollution.

1.3 METHODOLOGY

1.3.1 The Project Team

The objectives of this study were ambitious and demanded a wide range of expertise. The methodology that was followed was primarily strategic in nature, with carefully selected detail investigations. The project team structure conformed to this approach, in that selected specialists conducted the detail work according to circumscribed briefs, and strategic and scientific overview inputs were made by a small Core Team.

The Core Team comprised:

- Prof André Görgens, Sigma Beta and University of Stellenbosch – Project Leader
- Mr Nico Rossouw, Ninham Shand – Project Coordinator
- Prof Roland Schulze – University of Natal

The Specialists were:

- Prof John Annandale, University of Pretoria
- Mr Albert van Zyl, Institute for Soil, Climate and Water
- Prof Chris du Preez, University Free State
- Mr Ralph Meinhardt and E van der Walt, Plant Protection Research Institute
- Dr Nebo Jovanovic, University of the Western Cape
- Mr Bennie Grové, University Free State
- Dr Simon Lorentz, University of Natal
- Prof Martin Fey, University of Stellenbosch

1.3.2 Project Outputs

One or more colleagues who have complimentary skills and capacity supported the Specialists where necessary. A series of scientific assessments were done, on an individual pollutant basis, and each by a different Specialist, of the current state of knowledge, modelling capabilities and mitigation measures. These individual assessments were integrated by the Core Team to yield the Terms of Reference for a long-term research programme by the WRC.

The project also included a first order estimate of the contribution that agricultural activities make towards NPS pollution in South Africa. This was a high-risk challenge, given the constraints of time and budget that applied to this project.

The Terms of Reference (TOR) for the long-term research project was based on a Conceptual Framework formulated jointly by the project team in a Workshop setting. The detail of the draft TOR was the responsibility of the Core Team.

Output 1 - First order assessment

Output 1 comprised first order estimates of the contribution that agricultural activities were making towards NPS pollution in South Africa. The approach followed was to partition aggregated observed water quality loadings into Point and Non-Point components for selected constituents at selected sites in three representative river systems by application of the large-scale national databases of point discharges, return flows and agricultural land and water use that underlie the National Water Resources Strategy and the Internal Strategic Perspective Projects recently completed by the DWAF. The first order assessment was published as a separate Project Report¹ by the WRC.

Output 2 - Discussion Papers

Position papers were prepared for the five key NPS water quality concerns that were identified at the workshop on NPS impacts of Agriculture that was held in June 2002. The five position papers and the responsible team leaders were:

- | | |
|------------------------|--------------------------------|
| • Pesticides | (Team leader: Ralph Meinhardt) |
| • Nutrients | (Team leader: John Annandale) |
| • Sediments and metals | (Team leader: Albert van Zyl) |
| • Pathogens | (Team leader: Nico Rossouw) |
| • Economic instruments | (Team leader: Bennie Grové) |

It was agreed at the planning meeting that the subjects of Pathogens and Economic Instruments would be investigated in less detail than the other three subjects. For pathogens, it was argued that agriculture was probably not a major contributor when compared to other nonpoint sources such as dense settlements and urban runoff. The following was identified as the main components of the position papers:

- Brief overview of the extent of the problem within a national context and statements on data and information constraints.
- Description of the origins and sources of the problem: identify the main sources of the specific agricultural NPS problem.
- Description of the transfer/transport/transformation processes and mechanisms: assess the level of understanding concerning the processes operating at various scales from microscopic level up to catchment level and an identification of the knowledge gaps.
- Description of models that simulate the production, transport and fate of the pollutants: assess the predictive (modelling) ability that is available to address each of the specific agricultural NPS problems and identification of knowledge gaps.
- Mitigation measures/best management practises to address the problem: assess the state of mitigation measures to address the specific agricultural NPS problem and identification of knowledge gaps.

For the preparation of the position papers, three forms of information gathering were employed:

- literature reviews, including internet searches
- expert interviews with South African and international specialists in the NPS modelling and management domain, as well as in related disciplines, who are not members of the project team
- electronic correspondence with international researchers and practitioners in the NPS modelling and management domain, as well as in related disciplines.

¹ Cullis, J. (2005). First Order Estimate of the Contribution of Agriculture To Non- Point Source Pollution in South Africa. WRC Report No. 1467/2/05, Water Research Commission, Pretoria.

The information gathered was subjected to evaluation and synthesis through the development of Position Papers according to the allocation of responsibilities for components of deliverables agreed at the Project Planning Workshop.

Output 3 - Terms of Reference for a long-term research programme

In order to accommodate the WRC's need for a terms of reference for the long-term research project before the last quarter of 2003, the Research Project TOR Design Workshop was moved forward to 28 and 29 August 2003. At this Workshop the specialist teams presented their position papers, and together with the Core Team, formulate an outline of the TOR of the eventual long-term research project. It was envisaged that the Core Team would then consult with selected external role-players and stakeholders and compile the draft TOR. This process was extended into 2004 due to unavoidable delays during the last quarter of 2003 and the first quarter of 2004.

2. SUMMARY OF THE DISCUSSION PAPERS

2.1 FIRST ORDER ASSESSMENT OF THE CONTRIBUTION OF AGRICULTURE TO NON-POINT SOURCE POLLUTION IN SOUTH AFRICA

The objective of this component of the study was to make a First Order Estimate of the contribution that agriculture makes to non-point source pollution in South African rivers. The Breede, Middle Vaal and Mgeni catchments were selected as a representative sample of different agricultural practices in South Africa. Individual sub-catchments were identified within these catchments based on the location of water quality sampling sites and the degree of agricultural activity. Different NPS assessment techniques were considered and it was decided to use a bottom-up approach to calculate the mean monthly, seasonal and annual net Agricultural Non-point Source (ANPS) load for each of these sub-catchments based on observed water quality data. The net ANPS load is the net incremental pollution load coming off a catchment after known sources have been accounted for. Agriculture is considered in its broadest sense and includes irrigated, dry land and subsistence farming, as well as forestry plantations and any other unknown pollution sources such as increased load from rural settlements, transport infrastructure in the rural areas or other activities influencing stream-flow and/or water quality.

The study found that reliable data on observed pollution levels was very limited and insufficient to support a bottom-up approach. Of particular concern was the lack of information on known point sources, as the POLMON database had been corrupted for two of the three study catchments. In addition, it was necessary to make a number of assumptions for, amongst other things, natural background loads and urban wash-off loads that could potentially influence the results. A further complication was the natural removal of nutrients, which was not adequately accounted for in the calculation of net ANPS loads. The use of the observed water quality at a "least impacted" reference gauge to give an indication of the natural background load, was also found to be problematic, as "least impacted" did not necessarily imply "natural". A result of these assumptions was that a number of negative loads were calculated, particularly for the mean monthly net ANPS phosphate and nitrogen loads.

Despite the uncertainty in the calculated net ANPS loads, the results were sufficient to draw some initial conclusions. Agriculture, in its broadest sense, appeared to have a major impact on salinity loads, particularly in areas with a high degree of irrigation and natural saline geology, such as some of the sub-catchments in the Breede where the net ANPS salinity load accounted for between 60% and 75% of the observed load coming off a sub-catchment. In general nutrient loads tended to be dominated by point sources rather than agriculture, which resulted in negative net ANPS loads. There were some exceptions such as the Hex (Breede), Henley (Mgeni) and Erferis (Middle Vaal). The net

ANPS nitrogen load of 7kg/ha/a in the Hex sub-catchment is particularly high - as is the net ANPS phosphate load of 0.12 kg/ha/a in the Nagel sub-catchments. The contribution of agriculture to nutrient loads may have been more significant had the natural removal of nutrient loads from point sources been included in the calculations. It was also found that the net ANPS load was greatest during the wet season and in some cases, such as in the Breede, there appeared to be a "first flush" impact at the start of the wet season.

2.2 DISCUSSION PAPER 1: NUTRIENTS: AGRICULTURAL CONTRIBUTION, MANAGEMENT AND MODELLING

Overview of the extent of the problem

There are two nutrients mainly implicated in pollution of water resources. They are nitrogen (N) and phosphorus (P). High nitrate (NO_3^-) levels in drinking water can be hazardous to infants and livestock, and high P and N levels in surface waters can cause serious eutrophication problems that result in nuisance plant growth and water resource impairment problems.

Phosphorus is not very mobile in soil, and therefore instances of elevated P levels in groundwater are quite rare. On the other hand, nitrate is very mobile, and NO_3^- enriched groundwater does occur in certain regions of South Africa such as in the Gordonia district, Prieska area, Ghaap Plateau, Springbok Flats and the Crocodile River.

Nutrient enriched surface water systems may encounter serious problems like nuisance growth of algae and macrophytes (eutrophication), in turn leading to water quality deterioration, algal toxin production, taste and odour problems, oxygen depletion and decline of more desirable fish species. Agricultural and urban run-off, and wastewater effluent disposal play a great role in eutrophication. Generally the most severe eutrophication problems occur where the scale of anthropogenic nutrient inputs far exceed the natural environment's ability to assimilate the nutrients.

Origins and sources related to agriculture

Nitrogen and phosphorus balances were done at national, provincial and field scale. For example, on a national scale, it was found that that on an annual basis, N outputs exceeded the input by 59 000 tons in the period 1971/72 to 1980/81. Research on three different ecotopes in the Free State, showed that these soils are able to release 5 to 20 kg N ha^{-1} through mineralization from the top 200 mm of the profile annually, viz. 3 to 6% of their organic N. It was found that on an annual basis the P input exceeded the output with 89 000 ton for the period 1971/72 to 1980/81. The amount of P added by fertilizers was more than the P removed by the crops, probably to increase extractable soil P to optimum levels and maintain them there. Similar mass balances were done at provincial and field scale. It was concluded that these fairly crude nutrient balances were useful to indicate whether or not we are sitting on a "time bomb". It appeared from the field case studies, that dryland cropping on deep soils that provide a buffer, like we see in most of our maize producing areas, is unlikely to be a significant contributor to nitrate leaching. On the other hand, dryland production on shallow soils, like we see in the Western Cape, could be problematic. However, it should be borne in mind that the crude balances do not highlight those conditions under which potentially significant pollution may occur.

Transfer, transport and transformation processes and mechanisms in the soil-plant-atmosphere system were also discussed. The emphasis of these discussions was on soil, because it plays a fundamental role in the behaviour and fate of N and P in agricultural systems.

Modelling the production, transport and fate of nutrient pollutants

Many simulation models describing the dynamics of the N and P cycles have been developed. These have been mainly used to study nutrient balances of agricultural lands, but are now being extended to address the study of environmental pollution. Many dynamic processes involving these nutrients

depend on a supply of carbon, so their accurate simulation is not possible without a detailed description of the dynamics of carbon in soil organic matter. Several approaches to modelling N and P were reviewed, with models ranging from simple empirical models to the more complex mechanistic models. Some of the process oriented models included routines for nitrogen fixation, mineralization and immobilisation, nitrification, plant uptake, leaching, denitrification, ammonia volatilisation and ammonia fixation. Phosphorus modelling was also discussed briefly and this included descriptions of processes such as solution, physical absorption, chemical absorption, immobilisation and precipitation.

Mitigation measures

Designing efficient management programs for either N or P in cropping systems to sustain maximum productivity with minimum environmental impact, requires not only knowledge of the two cycles, but also of the soils, crops and fertilizers involved. Efficient N and P management programs, therefore, need to be derived individually for each field or set of fields of similar nature. The best way to minimize N and P non-point source pollution from cropping systems, is to focus on practices that favour high crop yields and reduce soil erosion. A number of these practices were listed. The development of a sound fertilization program was also discussed. Such a program takes care firstly of soil pH, and secondly of the availability of essential plant nutrients. A recent innovation that shows promise in reducing surface and subsurface flow of N and P from arable land is riparian buffer zones. Removal efficiency was also discussed briefly.

Information gaps and research needs

- It is essential that problem areas of non-point source nutrient pollution be identified, and the extent of the problem quantified.
- In modelling N and P many processes have to be described empirically because purely mechanistic approaches are either too poorly understood, or it is too difficult to parameterise such models.
- There is a general lack of expertise locally in modelling N and P balances, and there is a clear need to integrate nutrient balance models with locally used water balance models.
- The output from field nutrient and water balance models needs to be integrated and run on a catchment scale, with groundwater simulations, to assess the impact of different cropping systems or land uses on groundwater reserves.
- Rough nutrient balances need to be made for the irrigation and feedlot sectors, because most of the nutrients applied to crops and removed in the harvested products are concentrated in urban areas and feedlots by the consumers of these products.
- Guidelines for managing waste are at present very general, and they can be made much more rigorous by improving our understanding and ability to predict soil processes.
- There is a need to increase knowledge of gaseous losses of N under effluent irrigation through better understanding of the processes involved.
- There is a need for the soil science community to develop effective, inexpensive and assessable measurement techniques, and the establishment of performance criteria that indicate when management changes are required.

2.3 DISCUSSION PAPER 2: PESTICIDES

Overview of the extent of the problem

A comprehensive literature search was done for pesticides and the review included data from South Africa. The most commonly detected and/or studied pesticides in South Africa were chlorpyrifos and endosulfan (mentioned 11 times in the literature), azinphos-methyl (10 times), atrazine, deltamethrin and penconazole (4 times). The number of times a pesticide was reported on in the literature is a very rough indication of its usage. However, in all literature studies, strong motivation was given as to why certain pesticide species were studied rather than others. Most studies were done on surface waters

and there was generally lack of information on groundwater contamination. Impacts are not only caused by pesticides currently in use but also by pesticides that have been banned, but are characterised by prolonged persistence such as DDT. The review is concentrated on pesticide residue research that was conducted from about 1990 up to about 2003. There is a very little pesticide monitoring data available for South African water systems and it is thus difficult to access data that would provide an indication of fluctuation in residue concentrations over time. The review focussed on pesticides in the atmosphere that has not reached the target agricultural crops, pesticides that have been washed from the soil surfaces after deposition, and the mobility, behaviour and fate of pesticides in the soil. These three compartments (atmosphere, runoff and soil) are discussed in detail in the discussion paper.

Modelling pesticides

The review of the pesticides also focused on the three compartments. The concern about pesticides in the atmosphere was spray drift, that is spray that drift off the target crop. The review describes a number of approaches that were followed to describe and simulate spray drift. Although these models existed, there was concern that they mostly addressed near-field impacts (i.e. drift into neighbouring areas) and may not adequately describe catchment scale impacts. For the modelling part, the processes related to the behaviour and fate of pesticides were discussed. A number of comprehensive reviews and models used worldwide, and their features, were analysed. Based on this literature review, state-of-the-art models use the convection-dispersion equation to simulate water and pesticide fluxes in the soil profile as well as the mobile/immobile phase to account for preferential flow. This is coupled to sorption and decay of chemicals depending on temperature and soil moisture.

Information gaps and research needs

The knowledge gaps identified in the modelling review indicated that the following processes will need attention: pesticide runoff, volatilisation, adsorption/desorption, plant uptake, preferential flow, upscaling from field to catchment, adsorption and degradation in the intermediate vadose zone, chemical and microbial degradation as affected by environmental conditions, and photodecomposition. Model validation using reliable field data will be essential at different time and spatial scales. Some processes related to the behaviour and fate of pesticides, for example runoff, should be modelled in conjunction with sediment transport. Adsorption/desorption of pesticides, plant uptake and fluxes in the soil profile, including preferential flow, can be modelled in conjunction with nutrients (nitrogen, phosphorus) and heavy metals. Interactions between different pollutants should also be investigated.

The following recommendations emanated from the pesticide report:

- A mechanistic approach is required to accurately predict pesticide runoff.
- Modelling of volatilisation needs to be validated in field trials.
- Pesticide-specific sorption coefficients need to be determined in laboratory experiments in order to accurately describe the adsorption/desorption process.
- Models including a sound plant water uptake subroutine are preferred in order to accurately simulate pesticide uptake by plants.
- The convection-dispersion approach has a more scientifically sound background, but it requires more inputs compared to convection approaches, in particular if processes like volatilisation, plant uptake and biochemical decay are solved simultaneously. In general, the level of detail needed to model solute transport and pesticide leaching is uncertain and case specific. Selection of the model or approach should therefore be based on the objectives that one wants to achieve.
- There is need to develop and adapt existing models to simulate preferential flow.
- There is also need to develop and adapt existing models to simulate fluxes and processes occurring in the intermediate vadose zone in order to allow for extrapolation from field to catchment scale.

- The factors controlling the chemical and microbial degradation rate of pesticides (soil properties, weather conditions, properties of the specific pesticide etc.) need to be investigated. This can be done under laboratory conditions, but data need then to be extrapolated to field conditions.
- Photodecomposition at plant and soil surfaces needs to be validated in field trials, as there is evidence that it can contribute considerably to the pesticide balance.

2.4 DISCUSSION PAPER 3: SEDIMENT AND METAL POLLUTION AND RELATED ACTIVITIES

Sediment is the most widespread pollutant of surface water and has negative off-site effects on most water users. Sediment is not only a widespread pollutant of surface water, but is also the most important medium for transport of phosphorus, heavy metals, toxic substances and pathogens from agricultural land to surface waters.

Heavy metals are elements that have a high density and belong largely to the transition group of the periodic table. The elements that are generally considered include arsenic, cadmium, cobalt, chromium, copper, mercury, molybdenum, nickel, lead, selenium, vanadium and zinc. The non-point source contribution of heavy metals from agriculture to surface waters is low since the inputs of heavy metals to dryland agriculture are minor and most of the applied heavy metals to agricultural land are strongly bound to soil particles. Heavy metals as a NPS pollutant from agriculture was not discussed further.

Overview of the extent of the problem

Many South African rivers carry a naturally high suspended-solids load and permanently turbid water is noticeable in rivers in the former Transkei area, KwaZulu-Natal and in many reservoirs around the country. The most quoted South African sediment yield estimates are Midgley's figure of 363 million tonnes (300 t/km²/yr), Schwartz and Pullen's value of 233 million tonnes (190 t/km²/yr) and Rooseboom's estimate of 100-150 million tonnes (82-122 t/km²/yr), based on the sediment yield of main rivers. The World Map on Status of Human-Induced Soil Degradation shows that water erosion, largely induced by agricultural practices, affects more than 70% of South Africa's surface area. Measured values of soil losses from undisturbed veld range from 0.02 t/ha/yr to 0.75 t/ha/yr and from cropland values commonly range from 25 t/ha/yr to as high as 60 t/ha/yr.

Origins and sources

The major sources of sediment are natural erosion and land disturbances associated with urbanisation, mining, silviculture and agriculture. Although erosion rates associated with surface mining and urbanisation are much higher than from agricultural sources, these activities only occupy a small percentage of the land attributed to agricultural activities. Natural erosion and sediment generated from cropland, rangeland and forests are briefly discussed.

Sediment production and transport processes

Erosion processes take place at different scales. The **catchment** is the land surface that contributes water and sediment to any given stream network. The relevant erosion processes that occur at the catchment scale are interrill and rill erosion, gully erosion, stream channel erosion and flood plain scour. The routing of sediment through the catchment is an important process at the catchment scale. A small catchment consists of a stream channel that is linked with **hillslopes**. A hillslope is an area extending from the watershed of a catchment down to the stream channel and represents the spatial variation of topography, soil and land management patterns along the hillslope. The erosion processes that occur at the hillslope scale are, interrill and rill erosion, and gully erosion. The spatial distribution of hydrological response units and hence the spatial connectivity of runoff-producing areas is an important process determining the spatial extent of the erosion processes along the hillslope and the major sediment delivery processes to the stream channel. A hillslope consists of **land facets** along the

hillslope. A land facet is an area of homogeneous topography, soil and land management. The land facet scale represents the combined processes of rill and interrill erosion. On the upper parts of hillslopes, particularly on those of convex form, interrill erosion is the dominant erosion process. Rills are initiated at a critical distance downslope where overland flow becomes channelled and experience has shown that 5 m is about the minimum slope length that will adequately represent a rill system.

Modelling sediment production and transport

One of the key problems facing water resource managers is predicting fluxes of sediment in rivers. A number of models are being developed to better understand the complex erosion processes that occur in agricultural catchments, but are of necessity simplifications of reality. The discussion paper provides a comprehensive overview of the types of erosion models used (qualitative rule-based models, empirical models, conceptual models, and physically-based models), models that estimate the sediment yield from a catchment and simulates processes such as erosion from overland flow, channel and gully erosion and sediment routing through a catchment. The paper also provides a good overview of issues related to scale of modelling and some of the other techniques applied to erosion studies such as knowledge-based approaches, potential and hazard maps, unit area loading, and loading functions. The predictive section of the report is concluded with a description of efforts to integrate models at different levels by simple and complex means.

Mitigation measures

The strategies for erosion control are generally based on covering the soil to protect it from raindrop impact; increasing the infiltration capacity of the soil to reduce runoff; decreasing the erodibility of soil by improving aggregate stability; and increasing surface roughness to reduce the velocity of runoff. A number of erosion control measures are being described under the headings of crop and vegetation management, soil management, mechanical methods and hydrological management. Generally, preference is always given to crop and vegetation, and soil management measures. These are less expensive and deal directly with both the detachment (sediment production) and transport (sediment delivery) phases of erosion. They are also more easily fitted into an existing farming system.

Information gaps and research needs

The following knowledge gaps in the modelling ability were identified:

- A limitation in almost all the sediment yield models is that they do not consider shallow subsurface redistribution of water. Therefore, they are unable to identify hydrological active areas due to saturation excess flow or exfiltration of throughflow (interflow). The inclusion of interflow and saturation excess flow runoff generation routines in sediment yield models will provide the basic structure to predict the areas of potential gully initiation and will give the models greater versatility as predictive tools when linking with suitable routines that model gully initiation and development.
- Both conceptual and physically-based sediment yield models address mainly interrill (sheet) and rill erosion, whilst ephemeral and classical gully erosion are rarely included as a model routine. There is a particular need for modelling studies of gully erosion to predict impacts of agriculture on gully erosion rates at various temporal and spatial scales, and their impacts on the hydrological and sediment yield processes at the hillslope- and catchment-scale.
- The spatial and temporal distribution of sediment production and deposition are key processes for understanding sediment delivery from agricultural land to streams, and the transport of sediment from streams to large rivers and impoundments.
- Current conceptual catchment models are ideal to identify problem areas and problematic land uses in a catchment requiring more rigorous studies, but should be supported with more detailed field scale models in selecting best management practices and erosion control measures.

In the paper a number of issues were identified related to integrated modelling frameworks / systems for modelling NPS pollution of sediments and associated pollutants in agriculture from the field to the catchment scale.

Recommendations

- Establishing a limited set of representative catchments monitored for the purposes of non-point source processes conceptualisation, and model verification and development including, as far as possible, existing DWAF monitoring points. Monitoring of representative catchments on a regular basis, with reliable data acquisition and storage, should represent a nested system of a large catchment scale (10-1000 km²) - small catchment scale (0.5-10 km²) - hillslope and plot scale.
- Building simulation capability in current South African agrohydrological models, such as linking it with an instream hydraulic and sediment routing model, and by embedding a small catchment scale model that is able to identify hydrological active areas due to saturation excess flow and interflow.
- Deepening local experience in the application of comprehensive models with international standing.
- Evaluate models that consider interflow and saturation excess flow runoff generation routines and which provide the basic structure to incorporate gully erosion models/gully mapping.
- Evaluate gully erosion models.
- Explore the possibilities that remote sensing hold for cost-effectively mapping gullies and gully growth.

2.5 DISCUSSION PAPER 4: WATERBORNE PATHOGENS IN AGRICULTURE

Overview of the extent of the problem

This discussion paper provided a reconnaissance level summary of the knowledge base about waterborne pathogens in agriculture. It was decided to investigate waterborne pathogens in less detail than other NPS concerns because agriculture was not regarded as a major contributor when compared to other nonpoint sources of waterborne pathogens such as dense settlements and urban runoff. The principle pathogens of concern to agriculture are those that can cause the outbreak of waterborne diseases. These pathogens are characterised as:

- Organisms that are released into the environment in large numbers, or are highly infectious to humans and animals at low doses.
- They can survive and remain infectious for long periods or they are resistant to water treatment, and
- Some types of bacterial pathogens can multiply outside of a host under favourable environmental conditions.

From an agricultural nonpoint source perspective, the key waterborne pathogens of concern have been identified as:

- Protozoa (Giardia and Cryptosporidium)
- Bacteria that can infect multiple species (Escherichia coli, Campylobacter & Salmonella)
- Bacterial infections associated with humans (Shigella)
- Human viruses (e.g. Hepatitis)
- Cyanobacteria and other toxic algal blooms
- Helminths

These pathogens of concern are discussed briefly in the Pathogens discussion paper.

Origins and sources

In agriculture, water plays a major role in mobilising and transporting pathogens. Rainfall washes pathogens from faeces or vegetation into the soil or as surface runoff into surface water streams and rivers. Intensive livestock farming enterprises (feedlots, piggeries etc.) pose the greatest risk of pathogen contamination. Land application of sewage and sludge can also act as concentrated sources of pathogens and requires proper treatment, handling and application methods to minimize the infection of humans and animals. Domestic animals are also a source of pathogen contamination in urban areas. They are a potential reservoir of a number of pathogens including *Giardia*, *Cryptosporidium* and *Salmonella*. Lastly, wildlife also acts as pathogen reservoirs and their wastes enter surface water streams during rainfall runoff events. Waterfowl can also add to the pathogen loads in dams and rivers where they nest and feed.

Modelling pathogens

Most models of bacterial die-off are based on Chick's Law, a simulation of a simple first-order reaction in chemical kinetics. In the paper a brief review was done of models that can be used for catchment assessment studies and the models were classified into four main categories:

- Watershed-scale loading models (simple methods, mid-range models, detailed models)
- Field-scale loading models
- Receiving water models (Hydrodynamic models, steady-state water quality models, dynamic water quality models, mixing zone models)
- Integrated modelling systems

Mitigation measures

The overall goal of pathogen management in agriculture is to prevent or reduce pathogen movement into surface waters and to facilitate pathogen die-off. Mitigation measures are generally targeted at four control points:

- Import to the farm – Measures to control pathogen import to the farm is intended to prevent the initial infection of farm animals by these pathogens.
- Animal operation amplification/proliferation – measures to break the cycle of pathogen amplification or proliferation in the intensive animal feeding unit.
- Waste management – measures targeted at the storage of waste products and management of different waste streams on the farm, and
- Export from the farm – measures to constrain the export of pathogens from the farm in surface water streams and rivers.

Information gaps and research needs

- Quantifying the extent of the problem in South Africa –The spatial extent of agricultural sources of high pathogen loads appears to be largely unknown. There is a need to identify particular high-risk agricultural management practices and assess the extent to which these occur in specific catchments, contribute to pathogen loads in surface waters and, ultimately, result in waterborne disease outbreaks.
- Pathogen export parameters - There is a need to derive typical pathogen export parameters for South African conditions and farming practices.
- The ecology of waterborne pathogens – There is a need for understanding the ecology of waterborne pathogens from source, viability and survivability from the host through the entire water system.
- Impacts on riparian zones – There is a need to investigate the impacts of livestock grazing in riparian areas and the association with the outbreak of waterborne diseases in humans using those water sources for drinking water purposes.

- Wildlife and waterborne diseases – There is a need to investigate the impact of wildlife on waterborne diseases and impact this may have on rural communities and eco-tourism.
- Source identification – There is a need to identify of the sources of waterborne pathogens through pathogen “finger printing” techniques and assess their significance in the outbreak of waterborne diseases.
- Hazard identification – There is a need to quantify the concentrations of waterborne pathogens that are necessary to create an outbreak of waterborne diseases in a human population.

2.6 DISCUSSION PAPER 5: ECONOMIC CONTROL MEASURES

Defining non-point source pollution in economic terms

Nonpoint source (NPS) pollution of water with sediment, nutrients and other chemicals is regarded in economic terms as an agricultural externality. More specifically it is classified as a spatial externality emphasising the location of pollution. Spatial externalities exist because farmers have no incentive to consider the cost that pollution imposes on others. More specifically agricultural non-point source pollution can be classified as technological diseconomy (negative externality) since it causes a misallocation of resources and benefit conflict between the polluters and pollution victims.

Goals of economic NPS pollution control policies

The process of designing comprehensive policies for controlling NPS pollution consists of defining appropriate policy goals, choosing appropriate instruments and setting these instruments at levels that will achieve the goals at least cost. The ideal aim is to maximise the expected net benefits to society from pollution control strategies. However, this ideal aim is unachievable since the relationship between economic damage and NPS pollution is largely unquantified because no market for cleaner water exists. Although non-market valuation techniques can be used to provide estimates that are good enough to indicate that the benefits from water quality improvements are large and worth pursuing these estimates are generally not accurate enough to choose economically efficient levels of pollution. Due to the problems associated with quantifying economic damage from pollution, reducing expected damages from pollution, although desirable does not constitute a measurable policy goal. Instead, it is necessary to adopt alternative goals that are measurable and that are believed to reduce expected damages.

Policy instruments for NPS pollution control

After measurable NPS policy goals have been set, the next step is to set policy instruments at levels that will achieve the goals at least cost. Policy instruments (taxes, subsidies, standards) induce polluters to explicitly take the cost of pollution into account and can be applied either to farmers' actions (design based) or to the results of their actions (performance based).

Performance-based instruments refer to pollution emissions generated by farmers or the ambient water quality of receiving water. Ambient water quality is the preferred base because it is difficult to observe NPS pollution emissions resulting in prohibitively high monitoring cost. Using water quality as the performance base is advantages because it measures the actual observed pollution problem and allows the polluters freedom of choice on how to achieve a specific water quality standard. A disadvantage is that it measures group response resulting in inefficiencies to control pollution at least cost. **Design-based** instruments refer to instruments applied to producer's variable input use and production technology choices. There are two different types of designed based instruments namely expected runoff bases (simulated output from NPS pollution models) and instruments targeted more directly on input and technology choices. Expected runoff based instruments are aimed at reducing the pollutant load through the application of best management practices. Input and technology based instruments refer controlling inputs, management practices or landuse.

Cost effectiveness

Cost effective control will only occur under complete information. Furthermore one needs to use differentiated instruments to achieve cost effective control due to spatial heterogeneity. Thus, achieving cost effective control is difficult and unpractical. A practical alternative, although not cost effective is to apply instruments uniformly as a second best alternative. The cost effectiveness of second best alternatives can be improved if you use economic criteria to target pollution reduction responsibilities to areas in the catchment with the lowest control costs.

3. SUMMARY OF THE MAJOR CONCLUSIONS

The conclusions of the knowledge review formed the foundation for the development of a terms of reference for a long-term research programme. The following research needs were identified.

- Fundamental research was required to prioritise the primary NPS processes, to evaluate models that describe these primary NPS processes, to collect data at plot and field scale in at least two different climatic regions, and to use the data to verify the field scale models and select the most promising ones for the further phases of the project.
- Applied and predictive research was required to integrate the field scale models for the priority pollutants, to verify the integrated field-scale model using data collected previously, and where appropriate, improve the field scale model for the nutrient and pesticide pollutant groups. Applied research was also required on field scale NPS pollution control measures.
- Research into integrative tools was required to configure and verify NPS models at a catchment scale using data collected for the project and where appropriate, improve on the catchment scale NPS models. There was also a need to develop tools to develop an interface between field and catchment scale models.
- Management support research was required to test the field scale and catchment scale models on a case study and to assess economic trade-offs of NPS control measures.

4. TERM OF REFERENCE FOR A LONG-TERM RESEARCH PROJECT

The output of the scoping project was a terms of reference for a long-term research project of nonpoint source pollution from agriculture. The major components of the research programme are summarised below. The detailed description of the research programme follows after the last discussion paper of this report.

Title: Development of an integrated modelling approach to prediction of agricultural non-point source (NPS) pollution from field to catchment scales for selected agricultural NPS pollutants.

The general and specific objectives of the long-term research project are:

General objective:

Developing an integrated modelling approach to prediction of agricultural NPS pollution from field- to catchment-scale for selected NPS pollutants.

Specific objective:

1. Establishment of fate-of-NPS-pollutant and other requisite data sets at field-, laboratory-, and quaternary catchment-scale, suitable for model improvement, development and verification.
2. Improvement of understanding of on-farm NPS pollution control measures and their modelling requirements.

3. Establishment of an improved field-scale model for simulation of agricultural NPS pollution loadings for phosphorus, nitrogen, selected pesticides and sediments, as well as for simulating the beneficial impacts on nearby receiving waters of on-farm NPS pollution control measures.
4. Establishment of an improved quaternary catchment-scale model for simulation of agricultural NPS pollution loadings for phosphorus, nitrogen and sediments, as well as for simulating the downstream benefits of on-farm NPS pollution control measures at the larger scale.
5. Development of guidelines about model usage to examine economic trade-offs and feasibility of agricultural NPS pollution control measures at field- and catchment-scale.

The research activities of the long-term research project were grouped into four phases:

Phase One is aimed at fundamental research activities dealing with nitrogen, phosphorus, sediment and selected pesticides. These include:

- A description and prioritisation of primary NPS processes for production of each of N, P, sediment and selected pesticides from agricultural activities.
- A comparative evaluation of adequacy of the algorithms for the primary NPS processes in existing field-scale and catchment-scale water balance and water quality models.
- Collection of data sets of all priority processes relevant to each pollutant type, assembled in experimental plot-scale or field-scale fate-of-pollutant studies, including wash-off zone, root-zone, vadose zone and groundwater observations. This should be undertaken in two different bioclimatic regions of South Africa and should distinguish between dry-land (rain-fed) and irrigated agriculture.
- A comparative critique of the aforementioned models via fate-of-pollutant verification studies in typical soil profiles and supported by laboratory-based pollutant transport and leaching tests.
- Identification of and motivation for selection of the most promising field-scale model for each pollutant type for South African applications in the agricultural NPS field and a description of further development needs of the identified models.

Phase Two is focused on applied and predictive research outputs for nitrogen, phosphorus, sediment and selected pesticides, and includes:

- The development of a field-scale model for N, P and sediments. The model algorithms for the fate of the individual pollutants can be either integrated or closely-coupled.
- Configuration and verification of the field-scale model for N, P and sediments using the continuous data sets collected in Phase 1.
- Development of an integrated or closely-coupled field-scale model for selected pesticides and sediments. The water balance "engine" of this model should correspond closely with that produced for N, P and sediments.
- Configuration and verification of field-scale models for the selected pesticides, using the continuous data sets collected in Phase 1.
- Development and prioritisation of on-farm NPS pollutant control measures, based on existing knowledge, and descriptions of additional field-scale process modelling requirements and other functional improvements dictated by the need to model on-farm pollutant control measures.
- Improvement of the field-scale model for the selected nutrient and pesticide pollutant groups, taking into account the development needs described earlier.

Phase Three is aimed at research into integrative and interface predictive tools for nitrogen, phosphorus and sediment but not pesticides. These research activities include:

- The collection of continuous data sets of pollutant concentrations and loadings in streamflow and groundwater for two sets of nested gauged quaternary-size catchments (100 – 500km²) in two different bioclimatic regions of South Africa.
- Configuration of the catchment-scale models identified in Phase 1 using the above continuous data sets.
- Comparative critique of the configured catchment-scale models via fate-of-pollutant calibration and verification studies, based on the nested catchment data sets.
- Identification of and motivation for selection of the most promising catchment scale model for South African applications in the agricultural NPS field, and a description of further development needs of the identified model.
- Development and verification of an improved catchment-scale model for local use.
- Development of guidelines for the use of the field-scale models to help parameterise field-scale processes in the catchment-scale model.
- Development and testing of a graphical user interface (GUI) to provide a transparent environment for interchange of information between the selected field-scale model and the catchment-scale model, as well as between the current time-series management and GIS support tools of the respective models.

Phase Four is focussed on research activities to provide predictive support for the management of nitrogen, phosphorus and sediment and includes:

- A model-based case study, by use of the field-scale models, of the relative beneficial impacts at fine scales, of alternative on-farm NPS control measures on pollutant delivery to receiving streamflow and groundwater.
- A case-study, based on the two nested catchments, demonstrating downstream pollution mitigation by alternative individual or combined on-farm NPS control measures.
- A case study to conceptually demonstrate an appropriate methodology for evaluation of the economic trade-offs of agricultural NPS pollution control measures.
- A case study to quantify the economic tradeoffs of agricultural NPS pollution control measures, both at farm-scale and at catchment-scale.
- Development of guidelines for the use of the field- and catchment-scale models for the evaluation of the economic trade-offs of agricultural NPS pollution control measures.

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The research in this report emanated from a project funded by the Water Research Commission, entitled *Modelling non-point source pollution in agriculture from field to catchment scale: A scoping study*. The project team wish to thank the Commission for providing the funds that have allowed the team the opportunity to work in this field. We like would particularly thank Mr Meiring du Plessis, the Research Manager for this project, for his unfailing helpfulness and courtesy.

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The compilation of the knowledge review was a team effort and following team members authored chapters of this report:

Chapter	Authors
Discussion paper 1: Nutrients: agricultural contribution, management and modelling	JG Annadale Department of Plant Production and Soil Science, University of Pretoria CC du Preez Department of Soil, Crop and Climate Sciences, University of the Free State
Discussion paper 2: Pesticides – Overview of the extent of the problem and modelling	R Meinhardt (Agricultural Research Council) E van der Walt (Agricultural Research Council) C Reinhardt (University of Pretoria) N Jovanovic (University of the Western Cape) J Dabrowski (University of Cape Town) L Burger (Environmental Management Services CC)
Discussion paper 3: Sediments and metal pollution and related activities	AJ van Zyl Institute for Soil, Climate and Water, Agricultural Research Council
Discussion paper 4: Waterborne pathogens in agriculture	JN Rossouw Ninham Shand Consulting Services
Discussion paper 5: Economic non-point source pollution control policy	B Grové Department Agricultural Economics, University of the Free State
Terms of reference for the long-term research programme to predict agricultural non-point source pollution from field to catchment scales for selected agricultural NPS pollutants	AHM Görgens Sigma Beta

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INTRODUCTION TO THE KNOWLEDGE REVIEW

1. INTRODUCTION TO THE PROJECT

1.1 BACKGROUND AND MOTIVATION

It is increasingly recognised that non-point source (NPS), or diffuse, pollution plays a major role in the degradation of water quality; specifically with respect to salinity, eutrophication (nutrient enrichment), sediments, pathogens, pesticides and some heavy metals. It is furthermore, increasingly accepted that it is infeasible to properly manage water quality without addressing the contribution from non-point-sources. Consequently, attention is increasingly devoted to the quantification of NPS pollution and to identify means to control it cost-effectively at source.

Since most of the land area is utilised for agricultural activities, agriculture has both locally and internationally been implicated as a major source of NPS pollution. It is thus necessary to assess the contribution that the different agricultural activities make to the different manifestations of NPS pollution, to devise the means through which these can be controlled and to determine and predict the effect that control measures will have to reduce NPS pollution. During 2001 the WRC published the document, "A Guide to Non-Point Source Assessment" (Report TT 142/01, by GC Pegram and AHM Görgens), which aims to assist users in understanding their NPS assessment needs in a particular situation, and thereby to enable them to identify and apply appropriate NPS assessment techniques. This Guide development process was generic and did not include any direct model development and testing, nor did it deal with the agricultural sector specifically. Understanding the production, delivery, transport and use components of agriculture-derived NPS loadings of water resources and having a predictive ability about the fate of agriculture-related NPS constituents are discrete research themes that would enhance the usefulness of that "Guide", in the agricultural domain.

The contribution that irrigation activities makes towards the salinisation of water resources has been studied for quite some time and is currently still receiving attention. Other water quality issues of concern that are potentially aggravated by agricultural activities are eutrophication (through fertiliser leaching and wash-off from human settlements), sediments (as a result of erosion), pathogens (from intensive animal production units), pesticides (through the application of insecticides, fungicides and herbicides) and some heavy metals. Although agricultural activities that give rise to the latter water quality issues have been the subject of previous studies, our present level of knowledge concerning them is not as advanced as for irrigation-induced salinisation of water resources. In order to plan future research activities and, specifically, a project that will address those issues that require priority attention, it is necessary to take stock of the present situation with regard to NPS pollution and the existing knowledge base. The process of developing and scoping such a project should involve the major researchers, academics and professional practitioners in these fields in South Africa.

1.2 OBJECTIVES OF THE PROJECT

General:

- To involve major players active in this field in a scoping exercise and knowledge gap analysis in order to develop the terms of reference for a longer term project that would establish an integrated model for the prediction of agricultural non-point source (NPS) pollution from field to catchment scales for the major agricultural NPS pollutants.

Specific:

- Produce first order estimates of the contribution that agricultural activities are making towards NPS pollution.
- Assess the level of understanding concerning the processes operating at various scales from the microscopic level up to catchment scale, for the main sources of agricultural NPS pollution.
- Identify the linkages between pollutants and processes that can mitigate or exacerbate the impacts.
- Assess the range of mitigating measures that are available to address each of the major sources of agricultural NPS pollution, and
- Assess the predictive (modelling) ability that is available to address each of the major sources of agricultural NPS pollution.

1.3 METHODOLOGY

The objectives of this study were very ambitious, given its relatively limited time frame and budget, and demand for a wide range of expertise. Therefore, the methodology that was followed needed to be primarily strategic in nature, with carefully selected investments in detail. The project team structure conformed to this approach, in that selected Specialists according to circumscribed briefs conducted the detail work and that the strategic and scientific overview inputs were made by a small Core Team.

The Core Team comprised:

- Prof André Gorgens, Sigma Beta and University of Stellenbosch – Project Leader
- Mr Nico Rossouw, Ninham Shand – Project Coordinator
- Prof Roland Schulze – University of Natal

The Specialists were:

- Prof John Annandale, University of Pretoria
- Mr Albert van Zyl, Institute for Soil, Climate and Water
- Prof Chris du Preez, University Free State
- Mr Ralph Meinhardt and Etienne van der Walt, Plant Protection Research Institute
- Dr Nebo Jovanovic, University of the Western Cape
- Mr Bennie Grové, University Free State
- Dr Simon Lorentz, University of Natal
- Prof Martin Fey, University of Stellenbosch

One or more colleagues who have complimentary skills and capacity supported the Specialists, where necessary. A series of scientific assessments were done, on an individual pollutant basis, and each by a different Specialist, of the current state of knowledge, modelling capabilities and mitigation measures. These individual assessments were integrated by the Core Team to yield the Terms of Reference for a long-term research programme by the WRC.

Output 1 comprised a first order estimate of the contribution that agricultural activities make towards NPS pollution in South Africa. This was a high-risk challenge, given the constraints of time and budget that applied to this project.

The Terms of Reference (TOR) for the long-term research project was based on a Conceptual Framework formulated jointly by the project team in a Workshop setting. The detail of the draft TOR was the responsibility of the Core Team.

The following tasks and activities were undertaken:

1. Planning Workshop:

On the 11 and 12th of March 2003 the team members conducted a Planning Workshop. The Workshop comprised two sessions; the Project Planning Workshop was scientific-technical nature to refine the terms of reference of the scoping study, and the Team Meeting was administrative in nature:

- Project Planning Workshop aims: to refine the objectives and modus operandus for the project, to agree on a feasible approach for making first order estimates of the contribution that agricultural activities are making towards NPS pollution, and to formulate a mutually-acceptable approach to the Specialist Evaluations of NPS pollution knowledge gaps and research needs regarding process understanding, prediction and mitigation.
- Team Meeting aims: to get agreement from the project team on allocation of responsibilities for components of deliverables, to prioritise tasks, to set target dates and to agree on the allocation of components of the budget on a task basis. WRC representatives and reference group members participated in the Project Planning Meeting. Outputs from the two workshops were used to update the methodology section and budget allocation without modifying the overall aims or the total project budget.

2. First order assessment

Output 1 comprised first order estimates of the contribution that agricultural activities were making towards NPS pollution in South Africa. The approach followed was to partition aggregated observed water quality loadings into Point and Non-Point components for selected constituents at selected sites in three representative river systems by application of the large-scale databases of point discharges, return flows and agricultural land and water use that underlie the National Water Resources Strategy and the Internal Strategic Perspective Projects recently completed by the DWAF. The first order assessment was published as a separate report² by the WRC.

3. Position Papers

Position papers was prepared for the five key NPS water quality concerns that were identified at the workshop on NPS impacts of Agriculture that was held in June 2002. The five position papers and the responsible team leaders were:

- | | |
|------------------------|--------------------------------|
| • Pesticides | (Team leader: Ralph Meinhardt) |
| • Nutrients | (Team leader: John Annandale) |
| • Sediments and metals | (Team leader: Albert van Zyl) |
| • Pathogens | (Team leader: Nico Rossouw) |
| • Economic instruments | (Team leader: Bennie Grové) |

It was agreed at the planning meeting that the topics, Pathogens and Economic Instruments, would be investigated in less detail than the other three subjects. For pathogens, it was argued that agriculture was probably not a major contributor when compared to other nonpoint sources such as dense settlements and urban runoff. The following was identified as the main components of the position papers:

- Brief overview of the extent of the problem within a national context and statements on data and information constraints.

² Cullis, J. (2004). First Order Estimate of the Contribution of Agriculture To Non- Point Source Pollution in South Africa. WRC Report No. 1467/2/05, Water Research Commission, Pretoria.

- Description of the origins and sources of the problem: identify the main sources of the specific agricultural NPS problem.
- Description of the transfer/transport/transformation processes and mechanisms: assess the level of understanding concerning the processes operating at various scales from microscopic level up to catchment level and an identification of the knowledge gaps.
- Description of models that simulate the production, transport and fate of the pollutants: assess the predictive (modelling) ability that is available to address each of the specific agricultural NPS problems and identification of knowledge gaps.
- Mitigation measures/best management practises to address the problem: assess the state of mitigation measures to address the specific agricultural NPS problem and identification of knowledge gaps.

For the preparation of the position papers, three forms of information gathering were employed:

- literature reviews, including internet searches
- expert interviews with South African and international specialists in the NPS modelling and management domain, as well as in related disciplines, who are not members of the project team
- electronic correspondence with international researchers and practitioners in the NPS modelling and management domain, as well as in related disciplines.

The information gathered was subjected to evaluation and synthesis through the development of Position Papers according to the allocation of responsibilities for components of deliverables agreed at the Project Planning Workshop.

4. Research Project TOR Design Workshop:

In order to accommodate the Commission's need for a terms of reference for the long-term research project before the last quarter of 2003, the Research Project TOR Design Workshop was moved forward to 28 and 29 August 2003. At this Workshop the specialist teams presented their position papers, and together with the Core Team, formulate an outline of the TOR of the eventual long-term research project. It was envisaged that the Core Team would then consult with selected external role-players and stakeholders and compile the draft TOR. This process was extended into 2004 due to severe time constraints on the project leader during the last quarter of 2003 and the first quarter of 2004.

5. Information Evaluation and Synthesis Workshop:

The Position Papers that were circulated among the team members for comment were largely finalised during the last quarter of 2003. It was envisaged that at the Evaluation and Synthesis Workshop, that was scheduled for 29 and 30 January 2004, the final draft reports will be systematically "signed off" in which all team members, as well as selected external role-players will participate. Due to unavoidable delays during the last quarter of 2003 and the first quarter of 2004, this workshop did not take place. Instead it was agreed with the team that the Knowledge Review document (this document) would be distributed to all the team members for comment and that the Core Team would make the final editorial changes to the report.

1.4 OBJECTIVE AND LAYOUT OF THIS DOCUMENT

The position papers were finalised as discussion documents and are presented in this document. An extended executive summary was also prepared to introduce the key issues addressed in each document. Each discussion document was prepared as a stand-alone document with an introduction and it was concluded with a list of references specific to the particular document. The five discussion papers dealt with the topics of nutrients (Discussion paper 1), pesticides (Discussion paper 2), sediments and metal pollution (Discussion paper 3), waterborne pathogens (Discussion paper 4), and

economic control policy (Discussion paper 5). No attempt was made to integrate the documents. It was the purpose of the Terms of Reference, the last paper in this report, to provide a more integrated view of the problem of modelling non-point source pollution from field to catchment scale.

1.5 CONCLUSIONS

The conclusions reached for each of the knowledge review topics formed the foundation for the development of a terms of reference for a long-term research programme. The following needs were identified for a long-term research project:

- Fundamental research was required to prioritise the primary NPS processes, to evaluate models that describe these primary NPS processes, to collect data at plot and field scale in at least two different climatic regions, and to use the data to verify the field scale models and select the most promising ones for the further phases of the project.
- Applied and predictive research was required to integrate the field scale models for the priority pollutants, to verify the integrated field-scale model using data collected previously, and where appropriate, improve the field scale model for the nutrient and pesticide pollutant groups. Applied research was also required on field scale NPS pollution control measures.
- Research into integrative tools was required to configure and verify NPS models at a catchment scale using data collected for the project and where appropriate, improve on the catchment scale NPS models. There was also a need to develop tools to develop an interface between field and catchment scale models.
- Management support research was required to test the field scale and catchment scale models on a case study and to assess economic trade-offs of NPS control measures.

1.6 RECOMMENDATIONS

It was recommended that the Water Research Commission release the terms of reference for a long-term research project as a solicited proposal. Such a research programme would put the Commission in a position to provide the agricultural community in South Africa with the tools and knowledge to predict agricultural NPS pollution from field to catchment scale for nutrients, sediment and pesticides, to develop appropriate mitigation measures at farm and catchment scale, and to use economic trade-offs to select the most appropriate pollution control measures for different spatial scales.

DISCUSSION PAPER 1:

NUTRIENTS – AGRICULTURAL CONTRIBUTION, MANAGEMENT AND MODELLING

by

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NUTRIENTS – AGRICULTURAL CONTRIBUTION, MANAGEMENT AND MODELLING

1. EXTENT OF THE PROBLEM IN SOUTH AFRICA

1.1 NUTRIENT ENRICHMENT OF GROUND AND SURFACE WATERS

There are two nutrients chiefly implicated in pollution of water resources. They are nitrogen (N) and phosphorus (P). High nitrate (NO_3^-) levels in drinking water can be hazardous to infants and livestock (Tredoux, 1993), and high P and N levels in surface waters can cause serious eutrophication problems that result in nuisance plant growth and water resource impairment problems (Walmsley, 2000).

The negative impacts can therefore be on both ground- and surface waters. These will be discussed separately.

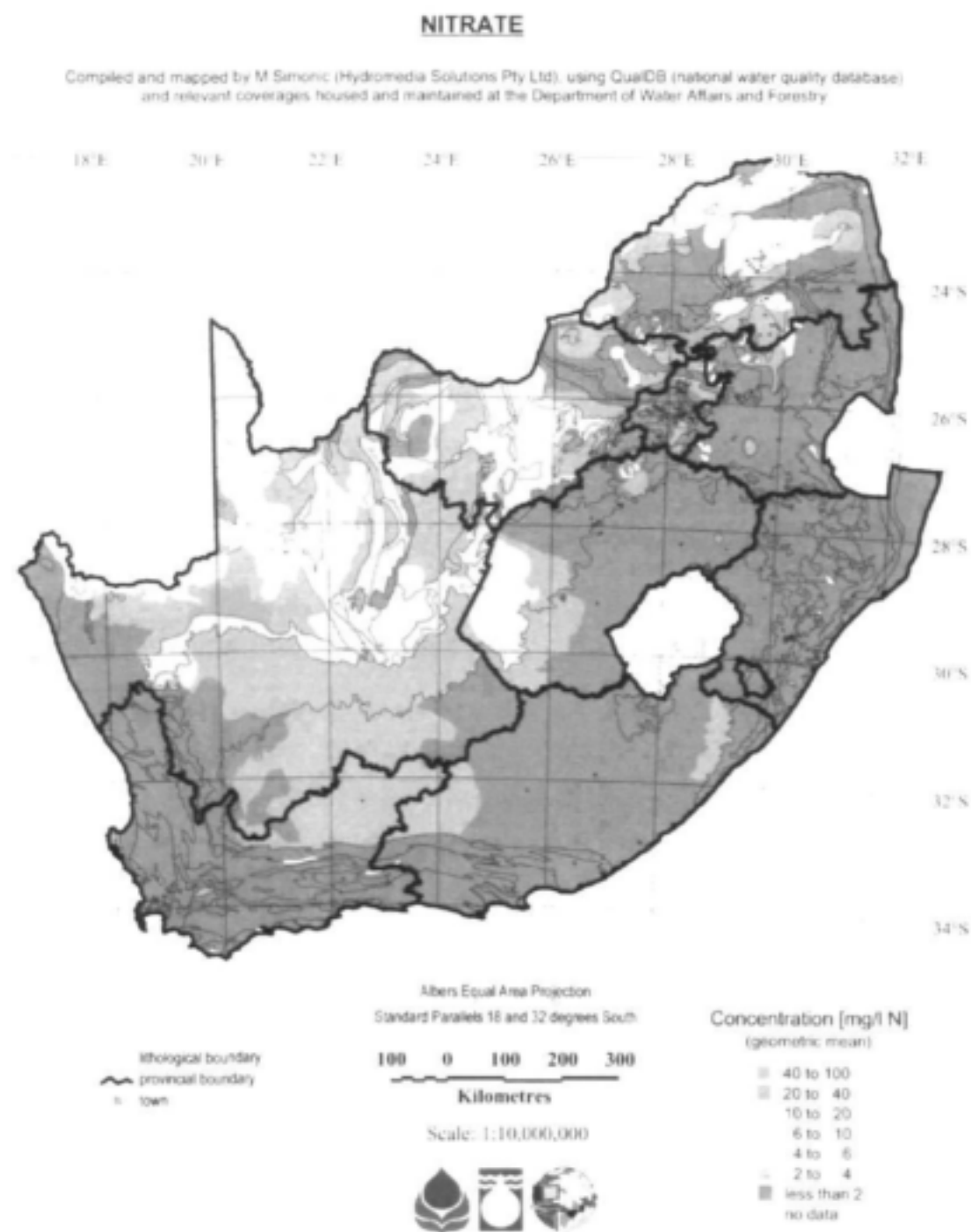
1.2 GROUNDWATER

P is not very mobile in soil, and therefore instances of elevated P levels in groundwater are quite rare. Only on very sandy soils subject to extreme P loading, is leaching likely to occur (Bond, 1998). This is therefore only important at certain waste disposal or effluent irrigation sites and feedlots. P levels in South African groundwater are very low (personal communication, Eddie van Wyk, DWAF groundwater monitoring), and are therefore not discussed further.

On the other hand, nitrate is very mobile, and NO_3^- enriched groundwater does occur in certain regions of South Africa. This can be seen in Figure 1.1 (Simonic, 1999).

Nitrate levels shown in Figure 1.1 have units of mg l^{-1} N, and not mg l^{-1} NO_3^- . Some confusion exists in the literature, as authors do not always clearly specify their method of reporting. In this report only N levels will be used unless otherwise specified.

Most authorities accept N levels below 10 mg l^{-1} (45 mg l^{-1} NO_3^-) for drinking water. At concentrations higher than this, methaemoglobinaemia or blue baby syndrome, may become a risk for infants (Colvin, 2000). Livestock are also affected, but at higher levels of nitrate. In 1977 the Namibian stock watering level was set at 110 mg l^{-1} (DWA Namibia, 1977). In South Africa, 90 mg l^{-1} was used as the limit for dairy cows (Baard, 1992), but water quality guidelines for livestock have been recently updated (DWAF, 1996) by Casey and Meyer (1996). No adverse effects are expected if nitrite N is below 10 mg l^{-1} and nitrate below 100 mg l^{-1} . Nitrate or nitrate poisoning is possible at nitrite levels exceeding 10 mg l^{-1} , but non ruminants and poultry appear to tolerate nitrate N levels up to 300 mg l^{-1} or nitrite N levels up to 100 mg l^{-1} . The possible effects of high nitrate concentrations on pregnant and non-pregnant ruminants and monogastrics have been highlighted by Casey and Meyer (1996). These authors also point out the conditions under which such waters may be tolerated.



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ASSESSMENT OF AMBIENT GROUNDWATER QUALITY ON A NATIONAL SCALE IN THE REPUBLIC OF SOUTH AFRICA
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Figure 1.1 Nitrate levels in South African groundwater (Simonic, 1999)

There has also been some concern that elevated nitrate levels may lead to an increased risk of stomach and oesophageal cancer, but the WHO (1985) has concluded that no firm link can be established due to the inadequacy of the available data.

Tredoux (1993) points out the regions of the country showing elevated nitrate levels in groundwater. These are in the:

- Gordonia District;
- Prieska vicinity;
- Ghaap Plateau, south-west of Vryburg;
- Springbok Flats, and
- Crocodile River.

The first two Tredoux (1993) considers examples of natural nitrate accumulation. The last two are caused by anthropogenic activities, whilst the Ghaap Plateaux is likely influenced by both natural and man made factors.

In Tredoux's (1993) thorough investigation into the nitrate concentration of groundwater in South Africa, he reports that at that stage the median NO_3^- concentration for the 18 827 sampling points was 4.5 mg l^{-1} , but that 27% of the sources exceeded 10 mg l^{-1} , 15% exceeded 20 mg l^{-1} and 4.3% exceeded 50 mg l^{-1} .

An accurate trend analysis of the NO_3^- contamination situation is essential in order to raise the alert as early as possible. This is especially important if one considers the slow response of many aquifers to changes in land use (Young, *et al.*, 1976; Colvin, 2000). Eddie van Wyk (personal communication) feels it is difficult to monitor local trends, as not enough sites are sampled regularly enough. Tredoux (1993) also highlights the fact that much data collected is of suspect quality, as not all laboratories and sampling methods have been up to standard in the past. The situation should be improving, with more and more laboratories receiving accreditation. Data gathering has also intensified over the past few years as a result of the regional groundwater mapping programme and the establishment of a national groundwater quality monitoring network (Simonic, 1999).

In his assessment of the ambient quality of groundwater in South Africa, Simonic (1999) states that the relatively short lifespan of national monitoring makes it very difficult to make authoritative statements on temporal trends. He makes several very useful suggestions as to how the national monitoring programme should be structured in future to get the maximum benefit at an affordable cost.

Simonic (1999) did attempt to do a temporal analysis on groundwater quality in the Springbok Flats. It was interesting to note that nitrate concentrations showed no significant change over a twenty year period (21 to $24 \text{ mg l}^{-1} \text{ N}$), and he suggests that this may indicate that NO_3^- changes are attributed to seasonal changes, rather than to a slow secular trend.

1.3 SURFACE WATER

Nutrient enriched surface water systems may encounter serious problems like nuisance growth of algae and macrophytes, in turn leading to water quality deterioration, algal toxin production, taste and odour problems, oxygen depletion, decline of more desirable fish species and in some cases, excessive loss of water through evapotranspiration (van Ginkel, 2002).

Water bodies range from *oligotrophic* (low in productivity and species abundance) through *mesotrophic* (moderate productivity and high species diversity) to *eutrophic* (high productivity and species abundance, but low species diversity). The extreme eutrophic condition is referred to as *hyper-eutrophic*. Causes of eutrophication can be natural, but are usually greatly accelerated by man's activities (Van Ginkel, 2002). In particular, agricultural and urban run-off, and waste water effluent disposal play a great role in eutrophication. Generally, the most severe eutrophication

According to Walmsley (2000), developed, industrialized countries use research and monitoring as an integral part of their strategies towards dealing with this problem. In particular, they are focussed on quantifying the nutrient sources, their impacts, and assessing the efficacy of policy and its implementation. These countries accept that cultural or man-made eutrophication is reversible, but there is also no quick fix. Transparent research and monitoring is prerequisite to the decision making that is required. Locally, Walmsley (2000) feels that the natural water resource monitoring system does not allow informed opinion as to whether the problem has increased or decreased in intensity. He feels a lack of input to policy development, monitoring, research, reporting and capacity building has greatly diminished the country's ability to deal with the problem.

The trophic status of selected South African impoundments (1990 - 2000)



Van Ginkel (2002) used the TN:TP (total N to total P) ratio to indicate the limiting nutrient. A TN:TP ratio above 10:1 signifies P limitation. From a eutrophication management perspective, she points out that this is desirable, as P is easier to manage and favours green algae, which may be less problematic to manage than cyanobacteria. From a eutrophication viewpoint, a TN:TP less than 10 represents an N limitation, which is less desirable as some of the cyanobacteria are able to fix

atmospheric N, and this favours their dominance in the system. Walmsley (2000) recognizes P as the fundamental cause of eutrophication, and points out that the systematic elimination of P in terms of its availability in surface waters is the only practical way of combating eutrophication.

Van Ginkel (2002) found 40% of the impoundment's sampling programs to be highly successful in providing data to determine trends in their trophic status. Figure 1.2 shows the whole range, from highly eutrophic (e.g. Hartebeespoort and Roodeplaat Dams) to those showing very little impact (e.g. Magoebaskloof and Sterkfontein Dams).

Van Ginkel (2002) ranked 10 impoundments needing urgent attention. These came from four catchments: The Crocodile/Pienaars River (Rietvlei, Klipvoor, Roodeplaat, Bon Accord and Hartebeespoort Dams); the upper Vaal River (Erferis and Bloemhof Dams); the Molopo River (Cook's Lake and Lotlamoeng Weir), and the Maas River (Shongweni Dam). She points out that the Crocodile/Pienaars River catchment experienced the highest level of eutrophication due to the extensive human activities within the upper reaches of the catchment, and all these impoundments are classed as hyper-eutrophic. As these are used extensively for human consumption and recreation, this catchment is in dire need of fast and effective management strategies. She also feels there is a need for a more extensive monitoring programme that should include all the country's impoundments, even those not experiencing problems at present so that a reference quality is available in future.

Walmsley (2000) argues strongly that immediate national research should be directed at quantitatively assessing the eutrophication problem in terms of its extent and trends, the sources of nutrients and the levels entering aquatic systems, and the actual social and economic costs of the problem on a national basis. Of course this would entail development of systems models for eutrophication management.

2. ORIGINS AND SOURCES OF THE PROBLEM

2.1 NITROGEN AND PHOSPHORUS BALANCES ON A NATIONAL, PROVINCIAL AND FIELD SCALE

2.1.1 National Scale - South Africa

Nutrient balances were calculated for both N and P by Biesenbach (1984) for all cultivated land in South Africa and some of the data is presented in **Table 2.1**. In the case of N, the inputs include fertilization and irrigation additions, while the outputs include crop and erosion removals. The author assumed that N additions from the atmosphere compensated for N losses by means of leaching, denitrification and volatilization. Since P is not susceptible to these losses he regarded atmospheric addition of P as an input. Other assumptions made by Biesenbach (1984) are not repeated here.

Inspection of **Table 2.1** shows that on an annual basis the N outputs exceeded the input by 59 000 tons from 1971/72 until 1980/81. However, it seems that this negative balance tended to decline over this period. For this 10 year period, the contribution to N inputs through fertilization averaged 310 940 t yr⁻¹, whilst irrigation contributed 610 t yr⁻¹. The annual contribution of crop uptake to N output averaged 329 970 tons, and erosion accounted for another 43 280 t yr⁻¹. Thus the amount of N added by fertilizers was 19 030 t yr⁻¹ less than the amount of N removed by crops.

According to Biesenbach (1984), from 1971/72 until 1980/81, 142 580 ton atmospheric N was added annually to the 14.2 million ha of cultivated land, viz. 10 kg ha⁻¹ yr⁻¹. Based on his assumption mentioned earlier, the same amount of N was considered lost through leaching, denitrification and volatilization. This N was probably released through mineralization of organic N. Research by Du Toit *et al* (1994) and Lobe *et al.* (2001) on three different ecotopes in the Free State, showed that these soils are able to release 5 to 20 kg N ha⁻¹ through mineralization from the top 200 mm of the profile annually, viz. 3 to 6% of their organic N.

The data of Biesenbach (1984) showed that on an annual basis the P input exceeded the output with 89 000 ton from 1971/72 until 1980/81 (**Table 2.1**). Over this period the positive balance tended to increase. For this 10 year period the atmospheric, fertilization and irrigation contributions to P input averaged around 11 400, 165 600 and 540 t yr⁻¹ respectively, while crop and erosion contributions to P output averaged around 54 200 and 34 200 t yr⁻¹ respectively. Thus the amount of P added by fertilizers was around 111 400 t yr⁻¹ more than the P removed by the crops, probably to increase extractable soil P to optimum levels and maintain them there.

Table 2.1 Nitrogen and phosphorus balances for cultivated land (14.2 million ha) in South Africa (Biesenbach, 1984)

Nutrient	Year	Input	Output ton x 1000	Balance
N	1960/61	48	219	-171
	1965/66	101	236	-135
	1971/72	209	373	-164
	1972/73	233	259	-26
	1973/74	234	412	-178
	1974/75	256	360	-104
	1975/76	299	312	-13
	1976/77	322	399	-77
	1977/78	347	375	-28
	1978/79	382	361	21
	1979/80	403	419	-16
	1980/81	469	458	11
	Average	314	373	-69
P	1960/61	76	59	17
	1965/66	113	65	48
	1971/72	144	87	57
	1972/73	155	68	87
	1973/74	174	94	80
	1974/75	169	85	84
	1975/76	171	79	92
	1976/77	176	92	84
	1977/78	182	93	89
	1978/79	190	86	104
	1979/80	203	96	107
	1980/81	212	104	108
	Average	178	89	89

2.1.2 Provincial Scale - North West Province

Bierman (2001) also calculated N and P balances using the same approach as Biesenbach (1984), but he focussed on the cultivated land of the North West Province. Some of these data are presented in Table 2.2. The only difference is that he was unable to account for N or P inputs through irrigation, but these should, in any event, be rather small.

Inspection of Table 2.2 shows that on an annual basis the N output exceeded input with 7 000 ton from 1994/95 until 1998/99. The balance was only positive in one of the five years, viz. in 1995/96. Over this five year period, the contribution of fertilization to N input averaged 42 460 t yr⁻¹, while the removal by crops averaged 45 760 t yr⁻¹, and erosion N output averaged 4 590 t yr⁻¹. Thus the amount of N removed by crops was 3 300 t yr⁻¹ more than the amount of N added by fertilizers.

Table 2.2 Nitrogen and phosphorus balances for cultivated land (1.5 million ha) in the North West Province (Bierman, 2001)

Nutrient	Year	Input	Output ton x 1000	Balance
N	1994/95	43	66	-13
	1995/96	38	35	13
	1996/97	49	59	-10
	1997/98	43	49	-6
	1998/99	40	42	-2
	Average	43	50	-7
P	1994/95	15	18	-3
	1995/96	15	10	5
	1996/97	15	16	-1
	1997/98	13	14	1
	1998/99	12	12	0
	Average	14	14	0

The amount of atmospheric N added to the 1.5 million ha cultivated land averaged 15 300 t yr⁻¹, viz. 9.8 kg N ha⁻¹ yr⁻¹ (Bierman, 2001). According to the assumption of Biesenbach (1984) mentioned earlier, the same amount of N was lost by means of leaching, denitrification and volatilization. As explained previously, this N was probably mineralised from organic N present in the soil.

On an annual basis from 1994/95 until 1998/99 the P input was generally equal to the P output. However, in 1995/96 input exceeded output by 5 000 ton, and in 1994/95 P output exceeded input by 3 000 tons. During this five-year period, the atmospheric and fertilization contributions to P input averaged around 1 220 and 12 650 t yr⁻¹, while the crop and erosion contributions to P output averaged about 10 220 and 3 670 t yr⁻¹. Thus the amount of P added by fertilizers was 2 430 t yr⁻¹ more than that removed by the crops, probably for the same reasons given previously.

2.1.3 Field Scale – Bethlehem, Malmesbury and Bloemfontein

Only N and P additions through fertilization and removals through grain were considered for three field scale case studies discussed here. Other additions and removals of N and P were omitted purposefully, as their quantification must be based on several assumptions due to a lack of measured data. However, this approach will show to some extent, whether fertilization is a large contributor to non-point source pollution of water or not.

Bethlehem

The mean grain yield of wheat over 18 years in a field trial was 2015 kg ha⁻¹ with no tillage, compared to 2145 kg ha⁻¹ with stubble mulch and 2204 kg ha⁻¹ with mouldboard ploughing (Table 2.3). However, grain yield varied from 0.3 t ha⁻¹ in 1994 to 5 t ha⁻¹ in 1993 mainly as a result of the amount and distribution of rainfall.

Table 2.3 Long-term effects of tillage practices on wheat grain yield and removal of nitrogen and phosphorus in the grain near Bethlehem. Yield data kindly supplied by Mr W H Kilian from the ARC-Small Grain Institute

Year	Yield (kg ha ⁻¹)			¹ Nitrogen (kg ha ⁻¹)			² Phosphorus (kg ha ⁻¹)		
	No tillage	Mulched	Ploughed	No tillage	Mulched	Ploughed	No tillage	Mulched	Ploughed
1979	1538	1413	1694	33.8	31.1	37.3	5.8	5.4	6.4
1981	2039	1980	2289	44.9	43.6	50.6	7.7	7.5	8.7
1982	853	1414	968	18.8	31.1	21.3	3.2	5.4	3.7
1983	1597	1779	1726	35.1	39.1	38.0	6.1	6.8	6.6
1984	2936	2838	2953	64.6	62.4	65.0	11.2	10.8	11.2
1985	918	908	1240	20.2	20.0	27.3	3.5	3.5	4.7
1986	3617	3354	3967	79.6	73.8	80.7	13.8	12.7	13.9
1987	2533	2671	2433	55.7	58.8	53.5	9.6	10.2	9.3
1988	2327	2318	2442	51.2	51.0	53.7	8.8	8.8	9.3
1991	1577	1571	1805	34.7	34.6	39.7	6.0	6.0	6.9
1993	4978	5177	5253	109.5	113.9	115.6	18.9	19.7	20.0
1994	315	357	285	6.9	7.9	6.3	1.2	1.4	1.1
1995	1226	1498	1690	27.0	33.0	37.2	4.7	5.7	6.4
1996	2774	2660	2868	61.0	58.5	63.1	10.5	10.1	10.9
1997	1218	1178	1501	26.8	25.9	33.0	4.6	4.5	5.7
1998	580	791	663	12.8	17.4	14.6	2.2	3.0	2.5
1999	2104	1953	2083	46.3	43.0	45.8	8.0	7.4	7.9
2000	3134	4781	4096	69.0	105.2	90.1	11.9	18.2	15.6
Mean	2015	2147	2204	44.3	47.2	48.5	7.7	8.2	8.4

¹ Annual nitrogen fertilization was 30 kg ha⁻¹ until 1998 and thereafter 40 kg ha⁻¹. Nitrogen content of grain was assumed to be 2.2%.

² Annual phosphorus fertilization was 13 kg ha⁻¹. Phosphorus content of grain was assumed to be 0.38%.

Irrespective of the tillage treatments, annual N fertilization was 30 kg ha^{-1} until 1998 and thereafter 40 kg ha^{-1} , which is more or less the recommended rate for commercial farmers. On average, more N was removed through the grain than was applied through fertilizer over the 18 year period. As can be expected from the variation in grain yield, in some years N was over applied, while in others, under fertilization occurred. The extremes were seen in 1993 and 1994.

The annual phosphorus fertilization was 13 kg ha^{-1} , which exceeded the average phosphorus removal through the grain, viz. 7.7 kg ha^{-1} for no tillage, 8.2 kg ha^{-1} for stubble mulch and 8.4 kg ha^{-1} for mouldboard ploughing. Phosphorus like N, was also over or under fertilized on an annual basis as a result of the variation in rainfall and hence grain yield. At present the extractable P content of this Avalon soil exceeds the optimum level, indicating that some of the fertilized P has accumulated in the soil.

Malmesbury

The average grain yield of wheat over 23 years in a field trial was respectively 1780, 1875, 1697 and 2064 kg ha^{-1} with no tillage (NT), minimum tillage (MT), conventional mulch tillage (CMT) and conventional clean tillage (CCT), as shown in Table 2.7. However, the grain yields varied from 114 kg ha^{-1} in 1978 (NT) to 5161 kg ha^{-1} in 1997 (CCT) as a result of a variation in the amount and distribution of rainfall.

On average far less N was removed through the grain (37.3 to $45.4 \text{ kg ha}^{-1} \text{ yr}^{-1}$) than was applied (60 to $100 \text{ kg ha}^{-1} \text{ yr}^{-1}$). Inspection of Table 2.7 shows that for most of the 23 years, excessive N was applied which will without doubt contribute to non-point source pollution. Nitrogen application rates of 60 kg ha^{-1} or more for wheat are very common in this region.

The average removal of P through the grain exceeded the P applied through fertilization. From the initiation of this long-term trial, the extractable P in this Swartland soil was above the optimum level.

Table 2.7 Long-term effects of tillage practices on wheat grain yield and removal of nitrogen and phosphorus in the grain near Malmesbury. Yield data kindly supplied by Prof G A Agenbag from the University of Stellenbosch

Year	Yield (kg ha ⁻¹)				¹ Nitrogen (kg ha ⁻¹)				² Phosphorus (kg ha ⁻¹)			
	NT	MT	CMT	CCT	NT	MT	CMT	CCT	NT	MT	CMT	CCT
1976	1028	875	707	884	22.6	19.3	15.6	19.4	3.9	3.3	2.7	3.4
1977	1125	902	323	1073	24.8	19.8	7.1	23.6	4.3	3.4	1.2	4.1
1978	114	151	147	253	2.5	3.3	3.2	5.6	0.4	0.6	0.6	1.0
1979	875	1018	868	990	19.3	22.4	19.1	21.8	3.3	3.9	3.3	3.8
1980	748	918	635	960	16.5	20.2	14.0	21.1	2.8	3.5	2.4	3.6
1981	414	865	677	1326	9.1	19.0	14.9	29.2	1.6	3.3	2.6	5.0
1982	716	711	541	652	15.8	15.6	11.9	14.3	2.7	2.7	2.1	2.5
1983	1306	1368	1180	1308	28.7	30.1	26.0	28.8	5.0	5.2	4.5	5.0
1984	1620	1654	1643	1849	35.6	36.4	36.1	40.7	6.2	6.3	6.2	7.0
1985	1826	1640	1558	1875	40.2	36.1	34.3	41.3	6.9	6.2	5.9	7.1
1986	2411	1940	1461	1676	53.0	42.7	32.1	36.9	9.2	7.4	5.6	6.4
1987	3789	2957	2888	3409	83.4	65.1	63.5	75.0	14.4	11.2	11.0	13.0
1988	2760	2249	1687	1306	60.7	49.5	37.1	28.7	10.5	8.5	6.4	5.0
1989	2808	3142	1552	3174	61.8	69.1	34.1	69.8	10.7	11.9	5.9	12.1
1991	2291	3506	3756	3727	50.4	77.1	82.6	82.0	8.7	13.3	14.3	14.2
1992	1104	1339	1450	1788	24.3	29.5	31.9	39.3	4.2	5.1	5.5	6.8
1993	1268	1885	1510	1929	27.9	41.5	33.2	42.4	4.8	7.2	5.7	7.3
1994	1290	1371	1199	1918	28.4	30.2	26.4	42.2	4.9	5.2	4.6	7.3
1996	3471	4186	4470	4613	76.4	92.1	98.3	101.5	13.2	15.9	17.0	17.5
1997	3988	4562	4824	5161	87.7	100.4	106.1	113.5	15.2	17.3	18.3	19.6
1998	1856	1979	1716	2201	40.8	43.5	37.8	48.4	7.1	7.5	6.5	8.4
2000	1839	1794	2106	2048	40.5	39.5	46.3	45.1	7.0	6.8	8.0	7.8
2001	2294	2108	2134	3341	50.5	46.4	46.9	73.5	8.7	8.0	8.1	12.7
Mean	1780	1875	1697	2064	39.2	41.2	37.3	45.4	6.8	7.1	6.4	7.8

¹ Annual nitrogen fertilization for all tillage treatments was 60 to 90 kg ha⁻¹ y⁻¹ until 1994 and thereafter 100 kg ha⁻¹ y⁻¹. Nitrogen content of grain was assumed to be 2.2%.

² Annual phosphorus fertilization was 4 kg ha⁻¹. Phosphorus content of grain was assumed to be 0.38%.

³ NT = no tillage, MT = minimum tillage, CMT = conventional mulch tillage and CCT = conventional clean tillage

Bloemfontein

The addition through fertilization, and removal through grain harvest, of either N or P from a field on a typical farm near Bloemfontein are presented in Table 2.4. From 1989 until 1996 which comprised five wheat crops, one maize crop and two fallow seasons, in total 214 kg N ha⁻¹ was removed in the grain, while in total only 113 kg N ha⁻¹ was fertilized. This is without doubt an indication of under fertilization of N and that the N reserves in this Bainsvlei soil were over exploited, resulting in organic matter decline. Over this eight year period, the P fertilized amounted to 69 kg ha⁻¹, while only 39 kg ha⁻¹ P was removed by the grain. Unfortunately, no data on extractable soil P is available to conclude whether excessive P was applied or not.

Table 2.4 Nitrogen and phosphorus addition through fertilization and removal through grain from a field on a typical farm near Bloemfontein. Yield and fertilization data kindly supplied by Prof A T P Bennie from the University of the Free State

Year	Crop	Yield (t ha ⁻¹)	¹ Nitrogen (kg ha ⁻¹)		² Phosphorus (kg ha ⁻¹)	
			Fertilizer	Grain	Fertilizer	Grain
1989	Wheat	1.20	18.0	26.4	12.0	4.6
1990	Wheat	1.46	16.0	32.1	10.0	5.5
1991	Wheat	1.32	14.0	29.0	11.0	5.0
1992	Fallow	-	-	-	-	-
1993	Wheat	1.30	15.0	28.6	11.0	4.9
1994	Wheat	0.65	14.0	14.3	10.0	2.5
1995	Fallow	-	-	-	-	-
1996	Maize	5.60	36.0	84.0	15.0	16.8

¹ Nitrogen content of maize and wheat grain was assumed to be 1.5 and 2.2% respectively.

² Phosphorus content of maize and wheat grain were assumed to be 0.3 and 0.38% respectively.

2.1.4 Conclusions – Nutrient Balance Studies

From the national and provincial scale nutrient balance studies, it seems that in comparison with crop removal, N was under applied and P was over fertilized. However, it is interesting to take note of the N and P use on South African cultivated land for two different ten year periods. The annual consumption from 1971/72 until 1980/81 was some 310 940 tons of N and 165 580 ton P (Biesenbach, 1984). From 1992 to 2001 the annual consumption was calculated from reports of the President of the Fertilizer Society of South Africa and found to be 395 000 ton N and 99 000 ton P. Thus the N consumption has increased and the P consumption has decreased over the past three decades, despite the fact that the area under cultivation remained more or less unchanged. Nowadays most cultivated soils have optimum extractable P levels and only maintenance fertilization is required. This is most likely why there has been such a marked decrease in P consumption. The increase in N consumption can be partly attributed to a decrease in the contribution of the now greatly depleted organic N, and hence less mineralizable N is now available as a result of cultivation (Du Preez and Du Toit, 1995). Product prices can also play a role.

Surveys of soils on commercial farms practising rainfed agriculture in South Africa, showed as in several other countries, large losses of N fertility (Du Preez & Du Toit, 1995). The pattern of these losses differs between ecotopes. An ecotope is defined as a region where the three environmental factors that affect yield, namely climate, slope and soil, are for practical purposes, homogeneous. The variation of these factors should not be sufficient to significantly influence the crops that can be produced, the yield potential of the crops and the production techniques.

In general, for all ecotopes, the declining pattern of N fertility owing to cultivation, showed three distinct phases (Du Preez and Du Toit, 1995). The rate of N fertility loss was rapid during phase I, which was in the first years of cultivation. Thereafter, the rate decreased during phase II until equilibrium was reached after several years of cultivation. After this, very little or no further loss occurred during phase III.

In the ecotopes Vryburg and Hennenman from the warmer, drier regions, an equilibrium in N fertility was approached much sooner than in the Tweespruit and Harrismith ecotopes from the cooler, wetter regions (Du Preez and Du Toit, 1995). The percentage N fertility loss was also greater in the wetter than in the drier ecotopes, viz. approximately 75% versus 50%.

This decline of soil nitrogen fertility has implications for non-point source pollution. During phase I, the rate of inorganic N release from organic N reserves is usually higher than the crop demand for inorganic N. This excess inorganic N, especially when converted to nitrate, can ultimately pollute water. The opposite is true during phase III, viz. the rate of inorganic N release from the organic N reserves is usually lower than crop demand for inorganic N. This insufficient inorganic N, especially in higher potential regions, increases the need for fertilization that may result in surface water pollution. Either of the two scenarios can occur during phase II.

The field scale nutrient balance studies indicated that when considering the removal of N and P through grain, both nutrients are often either over applied in some instances, or under applied in other cases. Both situations are detrimental to the environment as they result in degradation of soil and/or water quality. However, site specific fertilization recommendations are hampered by unpredictable climate and its influence on dryland crop yields, as was clearly illustrated by the three case studies.

These fairly crude nutrient balances are useful to indicate whether or not we are sitting on a "time bomb", as there can be very long time lags between a change in land use and an impact on groundwater. It seems from the field case studies, that dryland cropping on deep soils that provide a buffer, like we see in most of our maize producing areas, is unlikely to be a significant contributor to nitrate leaching. On the other hand, dryland production on shallow soils, like we see in the Western Cape, could be problematic.

On the other hand, these crude balances do not highlight those conditions under which potentially significant pollution may occur. For example, nitrate leaching is highly likely under intensively managed irrigated vegetable or other horticultural cropping systems, where water is added over and above the natural rainfall, and where economic returns are high, making fertilizer costs a relatively small fraction of the input costs, and farmers tend to over irrigate and fertilize as "insurance".

Also under dryland conditions, during poor cropping seasons, with an unfavourable rainfall distribution and minimal crop growth, conditions may also be conducive to nitrate leaching. This may seem counter intuitive, as rainfall is often below normal in such seasons, but because much of the applied N is not used by the poorly growing crop, it remains in the profile where it is vulnerable to loss through leaching should heavy rain occur.

The question is whether or not we have the tools and expertise locally to flag such scenarios? A wide variety of models are discussed later, but none have been used locally that we are aware of. It may be prudent to build capacity in this field locally, and perhaps even modify local crop growth and water balance models to include N and P subroutines in order to make such analyses. This would enable site specific and long-term simulations of the N and P dynamics of various cropping systems, that would enable informed decision making regarding the likely impact of these farming practices. Such models could also assist extension officers and fertilizer company agronomists to help farmers to

better manage their nutrients. We are dealing with "leaky systems" and even under the most careful management there will be some leaching. The challenge is to minimize the impact.

It is also clear that the nutrients in agricultural products are being concentrated in cities and feedlots, and these are then causing point source and non point source pollution problems. Future research should possible focus more closely on waste disposal sites like water care works that need to manage large volumes of sludge and effluent water. Feedlot waste handlers face similar problems. The nutrient and water balance models alluded to above, should prove invaluable in designing cropping systems to utilize waste products in as an environmentally friendly way possible.

3. TRANSFER / TRANSPORT / TRANSFORMATION PROCESSES AND MECHANISMS

3.1 CYCLING OF NITROGEN AND PHOSPHORUS IN SOIL-PLANT-ATMOSPHERE SYSTEMS

Nitrogen and phosphorus are nutrients as well as pollutants. An understanding of both the N and P cycles which are an integral part of the carbon cycle, is therefore of great importance for proper modelling of non-point source pollution in agriculture. Only the N and P cycles in the soil-plant-atmosphere system will be discussed concisely here, with most emphasis on the soil. In these cycles, organic matter is a vital component, since it contains approximately 95% of the N and 55% of the P found in soils. The N and P added to soils through fertilization is almost negligible in relation to the total amount of N and P in soils. Soils, therefore, play a fundamental role in the behaviour and fate of N and P in agricultural systems.

3.1.1 Nitrogen Cycle

The cycling of N in the soil-plant-atmosphere system is illustrated in Figure 3.1. This cycle can be divided into N inputs or gains, N outputs or losses, and N cycling within the soil, where N is neither gained nor lost by the processes listed in Table 3.1. Understanding the behaviour of these processes is essential for maximizing agricultural productivity and profitability, whilst reducing the impacts of N fertilization on the environment.

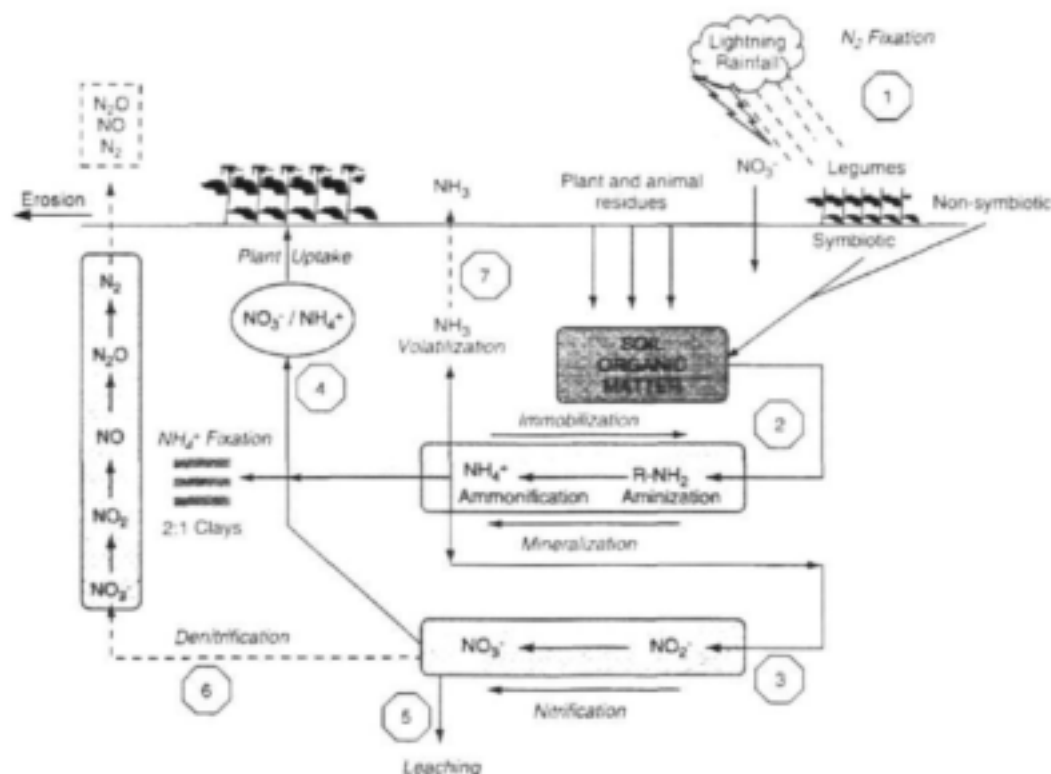


Figure 3.1 Nitrogen cycle in the soil-plant-atmosphere system (Havlin et al., 1999)

1. **Nitrogen fixation:** Nitrogen in plant and animal residues and N derived from the atmosphere through industrial, electrical and biological fixation is added to soil. Except for industrial fixation whereby nitrogenous fertilizers are manufactured, electrical and biological fixation occurs naturally. Usually, the nitrogenous fertilizers contain N in the form of urea, ammonium or nitrate. Electrical N fixation results through lightning, while biological N fixation is mediated by certain micro-organisms, which can be either free-living or symbiotic in nature. Symbiotic N fixation has a far greater effect on N availability to plants than non-symbiotic N fixation, although the biochemical processes involved with both are similar. Biological N fixation produces the primary flow of N in soil-plant-atmosphere systems with legumes, but is unimportant in the systems without legumes.
2. **Mineralization:** The organic N in the residues of plants and animals is mineralised to NH_4^+ by heterotrophic soil organisms. Some of the mineralised and fertilized NH_4^+ can be absorbed by plant roots, volatilised as NH_3 to the atmosphere, or adsorbed on the cation exchange complex of soils. The adsorbed NH_4^+ is not that susceptible to leaching, and some of it may even be fixed when soils have 2:1 clays. However, some of the NH_4^+ can also be immobilised into organic N by heterotrophic soil organisms. Mineralization and immobilization can occur simultaneously, but one process usually dominates the other, depending on the C/N ratio in the soil.
3. **Nitrification:** Most of the NH_4^+ that resulted from either fertilization or mineralization is oxidized via NO_2^- to NO_3^- by a small group of chemoautotrophic nitrifying bacteria. These bacteria keep concentrations of exchangeable NH_4^+ relatively low in aerobic soils, but their activity is inhibited by lack of O_2 in anaerobic soils. This nitrification process does not result in losses of N from soils, but does alter the susceptibility of N to loss through various processes, viz. plant uptake, leaching and denitrification. The NO_3^- that results from either fertilization or nitrification can also be immobilized into organic N by heterotrophic soil organisms.
4. **Plant uptake:** Uptake of NH_4^+ and NO_3^- by plant roots from soil is used to produce the protein in crops that are eaten by humans or fed to livestock. The overall process of N uptake is usually considered to involve movement to and across the plasma membrane of root cells as separate processes. Ammonium moves to the membrane primarily by diffusion, and NO_3^- often moves primarily by mass flow in the transpiration stream. Movement of these ions across the plasma membrane can occur by active transport and more than one mechanism may be involved. Rates of N uptake by plants tend to vary greatly with stage of growth. Many annual plants take up most of their N within a few weeks during the growing season.
5. **Leaching:** Leaching refers to the movement of N in water moving downward through the soil profile and out of the rooting zone. Nitrate is the form of N most likely to be lost by this process, because it usually is not adsorbed and hence the most abundant form of N in the water that moves. The importance of N losses by leaching varies greatly depending on the timing and magnitude of the flux of downward water movement in soils. Losses can be substantial in agricultural systems where fertilization or mineralization results in high concentrations of NO_3^- during periods when leaching is likely. In most soil-plant-atmosphere systems, losses of N by leaching are associated with short-term weather events that can be neither controlled nor predicted. Flows of water often increase N availability by moving soluble N from the soil surface to depths where roots are active, but this is usually not considered to be leaching. In many highly weathered soils, NO_3^- leaching can be retarded by positive charge present in acid subsoils, leading to a substantial storage capacity for NO_3^- in such soils.
6. **Denitrification:** Denitrification is the reduction of NO_3^- and sometimes also NO_2^- to nitrogen oxides and eventually N_2 by denitrifying bacteria. Conditions conducive to denitrification

include absence of O_2 , and presence of NO_3^- or NO_2^- and an electron-donating substrate like organic residues to support microbial respiration. Although the absence of one or more of these conditions usually prevents rapid denitrification, a short period of favourable conditions can result in substantial losses of NO_3^- from soil.

7. **Ammonia volatilization:** Ammonium in soil solutions tends to equilibrate with NH_3 in air near the solutions, and this equilibration often results in losses of N through NH_3 volatilization. Such losses are favoured by high pH and short distances between the solution and moving air. Significant losses from soils are most likely to occur when NH_4^+ from fertilizers or animal manure remain on the surface. Plants can also lose N as NH_3 through stomata. Soils and plants at other sites may absorb some of the NH_3 volatilised from one site, and such absorption may provide a source of plant available N under some conditions.
8. **Ammonium fixation:** Certain clay minerals, particularly vermiculite and illite, are capable of fixing NH_4^+ by a replacement of NH_4^+ for interlayer cations in the expanded lattices of clay minerals. The fixed NH_4^+ can be replaced by cations that expand the lattice (Ca^{2+} , Mg^{2+} , Na^+ , H^+) but not by those that contract it (K^+). Fixation of freshly applied NH_4^+ can occur in all particle size fractions if they contain substantial amounts of vermiculite. The availability of fixed NH_4^+ ranges from negligible to relatively high.

All the processes involved with cycling of N in the soil plant-atmosphere system are reviewed in much more detail elsewhere: Nitrogen fixation (Havelka *et al.*, 1982; Stevenson, 1982; Haynes, 1986a), mineralization and immobilization (Jansson and Persson, 1982; Haynes, 1986b), nitrification (Schmidt, 1982; Haynes, 1986c), ammonium fixation (Nommik and Vahtras, 1982; Cameron and Haynes, 1986), leaching (Nielsen *et al.*, 1982; Cameron and Haynes, 1986), denitrification (Firestone, 1982; Haynes and Sherlock, 1986) and plant uptake (Olsen and Kurtz, 1982; Haynes, 1986d). Some of these processes can be influenced through soil and crop management, but generally are not easy to manage (Havlin *et al.*, 1999).

Table 3.1 Nitrogen inputs, outputs and cycling in the soil (Havlin *et al.*, 1999)

Inputs / Gains	Outputs / Losses	Cycling
Fixation	Plant uptake	Mineralization
Industrial	Denitrification	Immobilization
Electrical	Volatilization	Nitrification
Biological	Leaching	Ammonium fixation
Plant residues		
Animal manure		

3.1.2 Phosphorus Cycle

The cycling of P in the soil-plant-atmosphere system is conceptualised in Figure 3.2. As in the case of the N cycle, the P cycle can also be divided into P inputs or gains, P outputs or losses and P cycling within the soil where P is neither gained nor lost by the processes listed in Table 3.2. An understanding of these processes is essential to ensure a basis for management of this element for optimum agricultural production with minimal environmental damage.

Gains of P in soils are attributed to fertilizers and plant or animal residues. Commercial fertilizers such as super- and ammoniumphosphates, contain mainly inorganic P compounds, which are water soluble. However, sometimes water insoluble inorganic P compounds are added to acid soils as rock phosphate. Mainly organic P compounds occur in either plant or animal residues that have been identified primarily as inositol phosphates, phospholipids and nucleic acids.

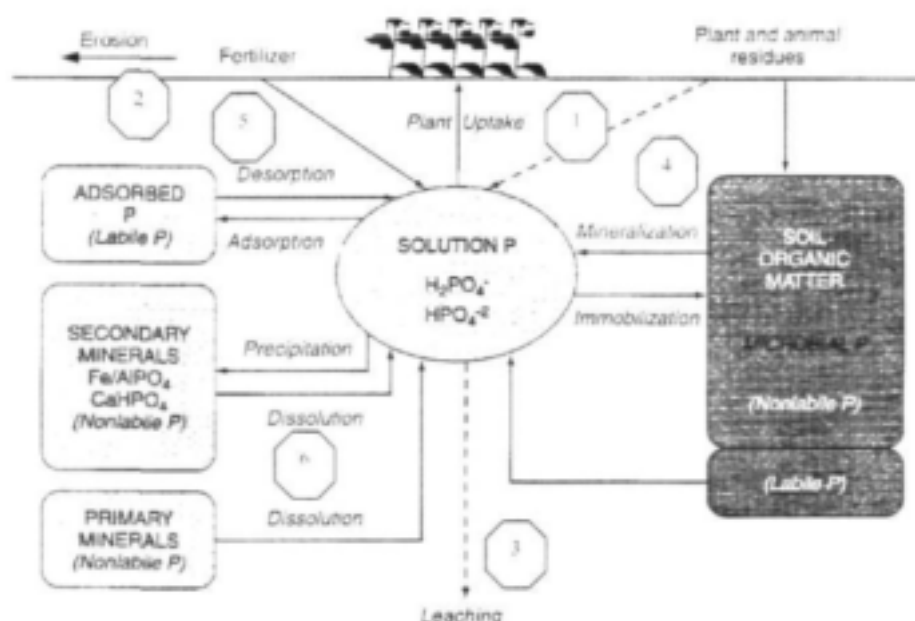


Figure 3.2 Phosphorus cycle in the soil-plant-atmosphere system (Havlin *et al.*, 1999)

Losses of P from soils are attributed to plant uptake, erosion and leaching.

1. **Plant uptake:** Plants obtain P from the soil solution actively in the form of H_2PO_4^- and HPO_4^{2-} ions. However, plant uptake of HPO_4^{2-} appears to be slower than that of H_2PO_4^- due to the use of different carriers. Both ions move to the roots by diffusion. In soils with a pH below 7 the dominant ion is H_2PO_4^- while in soils with pH above 7, the dominant ion is HPO_4^{2-} .
2. **Erosion:** Large losses of the various P fractions occur through sediments as a result of either water or wind erosion. Both H_2PO_4^- and HPO_4^{2-} are strongly sorbed to soil particles when incorporated by means of a fertilizer in the topsoil. The sorption processes are explained later.
3. **Leaching:** For both ionic forms, leaching is limited due to their easy transformation into immobile forms. Therefore H_2PO_4^- and HPO_4^{2-} are unlikely to move to groundwater except on very sandy soils under excessively high loading rates of P, like is sometimes found with effluent irrigation and waste disposal.

Cycling of P within the soil involves several biological and chemical processes that influence the soil solution P concentration. The biological and chemical processes will be discussed separately to indicate how the P concentration in the soil solution is buffered by the various organic and inorganic P fractions. The biological processes comprise mineralization and

immobilization, whilst the chemical processes are adsorption and desorption as well as precipitation and dissolution.

4. **Mineralization / immobilization:** These biological processes comprise mineralization of organic P to inorganic P and immobilization of inorganic P to organic P by heterotrophic soil organisms. In general these two processes are similar to those for N in that both occur simultaneously with one usually dominating the other depending on the C/N ratio in the soil. Several other factors also effect the direction, extent and rate of these processes in soil.
5. **Adsorption and desorption:** The amount of P sorbed is controlled by the soil solution P concentration. However, for a given solution P concentration there is a large difference in the amount of P sorbed by different soils. In addition P sorption is also a function of the amount and type of hydrous oxides of Al and Fe and reactive Ca compounds present, other ions, pH of the system, and reaction kinetics. The release of sorbed P into solution, viz. desorption is usually not completely reversible, since all that has been sorbed cannot desorb.
6. **Precipitation and dissolution:** The solubility product of the least soluble P component in the solid phase controls dissolution and thus solution P concentration. In general, Ca controls these reactions in neutral or calcareous soils, while Al and Fe are the dominant controlling cations in acidic soils. As a result, several secondary phosphate minerals are formed in soils and these minerals vary widely in solubility that results in different dissolution rates. Apatite is the most common primary P mineral and its dissolution requires a source of H⁺ from soil or biological activity, and a sink for Ca and P. The dissolution of apatite varies with rainfall and temperature and is therefore quite difficult to model.

The P cycle can be simplified to soil solution P ↔ labile P ↔ non-labile P where labile and non-labile P represents both inorganic and organic fractions. Labile P is the readily available portion of both fractions that exhibits a high dissociation rate and rapidly replenishes soil solution P. Depletion of labile P causes some non-labile P to become labile, but at a slow rate.

All these P cycling processes in the soil-plant-atmospheric system are reviewed in much more detail by Stevenson and Cole (1999). Detailed reviews on the chemical processes controlling P availability have also been presented by Barrow (1985), Syers and Curtin (1989), and Frossard *et al.* (1995).

Table 3.2 Phosphorus inputs, outputs and cycling in the soil. (Havlin *et al.*, 1999)

Inputs / Gains	Outputs / Losses	Cycling
Fertilization	Plant uptake	Mineralization / Immobilization
Plant residues	Erosion	Precipitation / Dissolution
Animal manure	Leaching	Adsorption / Desorption

4. MODELLING THE PRODUCTION, TRANSPORT AND FATE OF NUTRIENT POLLUTANTS

Many simulation models describing the dynamics of the N and P cycles have been developed. These have been mainly used to study nutrient balances of agricultural lands, but are now being extended to address the study of environmental pollution (McGechan and Wu, 2001). Many dynamic processes involving these nutrients depend on a supply of carbon, so their accurate simulation is not possible without a detailed description of the dynamics of carbon in soil organic matter.

Several approaches to N and P modelling will be briefly reviewed below, with reference made to more comprehensive reviews.

4.1 NITROGEN MODELLING

An excellent, up to date publication on this topic, has been produced by Shaffer *et al.* (2001). Their multiple author monograph entitled "Modeling Carbon and Nitrogen Dynamics for Soil Management", covers reviews of several models and applications of many of these models at field and catchment scales.

Shaffer *et al.* (2001) also discuss different levels of complexity of the various models, adding that with C/N cycling models, there has been a strong tendency to develop mechanistic models that attempt to describe the various biological, chemical and physical mechanisms at play. Some models like *ecosys* (Grant, 2001) are research oriented with very detailed descriptions of the processes simulated. Others like NTRM (Shaffer and Larson, 1987) are applications oriented, whilst models like NLEAP (Shaffer *et al.*, 1991) can be used as screening tools. The scale of application ranges from short term seasonal estimates like NLEAP, to long-term trends in carbon sequestration as in CENTURY (Parton *et al.*, 1996).

Shaffer *et al.* (2001) point out that from a spatial viewpoint, the more complex research models tend to be point based, while broad scale regional modelling is left to simpler screening tools. The recent increase in computing power has resulted in linkages of process-based models to GIS, and this has changed somewhat the definition of research vs. application tools. However, Garnier *et al.* (1998) remind us that the availability of data to support research models across broad regions continues to be problematic. Shaffer *et al.* (1991) recommend development of generic parameter databases for models across a wide range of soil and land management scenarios to extend the usability of the models, but such databases do not exist for most models.

C/N models are difficult to parameterise, especially as several pools are considered (Shaffer *et al.*, 2001). Pool sizes and associated rate parameters are difficult to measure, and often require some model pre-runs for self-parameterisation and initialisation (Ma *et al.*, 1998). Parton *et al.* (1994) highlight the fact that numerous attempts have been made to develop field and laboratory methods to quantify N pools, but these have not been that successful.

Hansen *et al.* (1995) indicate another difficulty with C/N models is determining the rate coefficients used for various process simulations. These may be zero- or first-order, Michaelis Menten kinetics, or Monod kinetics. The rate coefficients may be constants, but are often functions of soil water, temperature, pH and O₂. Most models use a zero to one response effect for each environmental factor, but there is a trend to use integrated response curves for a combination of factors (Shaffer *et al.*, 2000).

Shaffer *et al.* (2001) also warn of the need to take into account the importance of correct parameterisation of other model components like for water and solute movement, plant growth and heat flow, as errors here will directly influence the C/N simulation and parameterisation. Another issue they raise is one of scale. Even if a field scale model is exceptionally accurate at the scale for which it was developed, it may still fail when used at a different, e.g. catchment, scale. They also feel that a lack of sufficiently accurate temporal and spatial field measurements for model development, calibration and testing of models remains problematic.

As far as the accuracy of the current C/N models is concerned, Shaffer *et al.* (2001) indicate that comparisons of model output to measured data show that the general trends are well simulated with some intermediate details, but the finer details are not being adequately reproduced. This statement is true for both research and application level models, and there is clearly room for improvement. They also advise some degree of regional or local calibration before attempting extrapolations in time or space.

Although farmers can readily be shown the impacts of their management decisions on the environment, and given answers to what-if questions regarding crop response to N, leaching of nitrates, and greenhouse gases, it is very unlikely that they will be direct users of such models. For this reason, Shaffer *et al.* (2001) recommend the need for model based decision support databases when it comes to extension and technology transfer. Finally, they allude to the keen interest there is in using models as regulatory tools. In such cases, model calibration and testing prior to their use in this arena is essential.

4.2 N MODELS

The models considered in the reviews of Ma and Shaffer (2001) and McGeachan and Wu (2001) are:

ANIMO, developed by the Winand Staring Centre, Wageningen, the Netherlands (Berghuijs-van Dijk *et al.*, 1985);

CANDY, from the Leipzig-Halle Environmental Research Centre, Germany (Franco *et al.*, 1995);

CENTURY, Colorado State University, USA, (Parton *et al.*, 1994);

CERES, Crop Estimation through Resource and Environment Synthesis, University of Hawaii, USA (Hanks and Ritchie, 1991);

DAISY, from the Royal Veterinary and Agricultural University, Denmark (Hansen *et al.*, 1991);

EPIC, Erosion/Productivity Impact Calculator, Texas A&M University, USA (Williams, 1995);

GLEAMS, Groundwater Loading Effects of Agricultural Management Systems, University of Georgia, USA (Leonard *et al.*, 1987);

LEACHM, Leaching Estimation and Chemistry Model, Cornell University, New York, USA (Hutson, 2000);

NC SOIL, University of Minnesota, USA (Molina *et al.*, 1983);

NLEAP, Nitrogen Leaching and Economic Analysis Package, USDA-ARS, Fort Collins, Colorado, USA (Shaffer *et al.*, 1991);

NTRM, the Nitrate, Tillage and crop-Residue Management model, USDA-ARS, Great Plains Systems Research Unit, Colorado, USA (Shaffer and Larson, 1987);

RZWQM, the Root Zone Water Quality Model, USDA-ARS, Great Plains Systems Research Unit, Colorado, USA (Ahuja *et al.*, 2000);

SOILN, set up by the Swedish University of Agricultural Sciences, Sweden (Johnson *et al.*, 1987);

SUNDIAL, the Rothamstead Nitrogen Turnover Model (Bradbury *et al.*, 1993)

No Australian models were considered in the reviews of Ma and Shaffer (2001) and McGeachan and Wu (2001), but **APSIM** is likely the most important model that deserves attention. We are not aware of any locally developed field scale N balance models, and ACRU, a local catchment scale model does not consider nitrogen either (Dr Lorentz, personal communication).

All the models considered, simulate multiple components in agricultural systems, with varying levels of complexity and success (Ma and Shaffer, 2001). They also point out that although all the models have achieved various degrees of success, they all have their weaknesses and will fail under certain circumstances. Few authors specify clearly the limitations of their models.

Ma and Shaffer (2001) point out that EPIC, developed at Texas A&M, has been modified to simulate several fields in a watershed by adding channel flow. This model is called APEX (Agricultural Policy Environmental Extender). They also add that EPIC and GLEAMS are very similar in their simulation of soil nutrient dynamics (N and P).

Ma and Shaffer (2001) note that the CENTURY model was developed to simulate long-term effects (minimum 10 years) on soil and carbon dynamics on a monthly time step. The model has recently been modified to handle a daily time step (DAYCENT), and also includes N gas flux from the model of Li *et al.* (1992). Ma and Shaffer (2001) point out further that CENTURY has been used to simulate carbon and nutrient (N, P and S) dynamics of several different ecosystems like grasslands, agricultural fields, forests and savannas. As the model uses a monthly time-step and is targeted at long-term carbon sequestration, the N sub-model is simple, with no differentiation between ammonium and nitrate, so nitrification, denitrification, ammonia volatilization and urea hydrolysis are not simulated.

Ma and Shaffer (2001) indicate that a unique feature of NCSOIL is that it has the capability of simulating N and C tracers simultaneously with non-traced soil organic C and N to study the mechanisms of N and C cycling. The model can also be used as a stand-alone model.

NLEAP was developed by Shaffer *et al.* (1991), as part of a national (USA) effort to quantify mechanisms for nitrate leaching from agriculture. This model has been widely used for water quality assessment, and a Java based Internet version has now been developed with GIS capabilities and linkages to NRCS (Natural Resources Conservation Services) databases (Shaffer *et al.*, 2001). NLEAP has been applied in studies identifying nitrate leaching at a regional scale (Shaffer *et al.*, 1995).

Ma and Shaffer (2001) describe RZWQM as a process level simulation model of soil water, soil temperature, plant growth, pesticide fate, and C and N dynamics. The most important feature of this model is that it is able to simulate in great detail, the effect of various agricultural management practices on water quality and crop production. It also simulates macropore flow and water table fluctuations.

Ma and Shaffer (2001) explain that NTRM preceded NLEAP and RZWQM. The complexity of C and N processes increases therefore from NTRM to NLEAP and then to RZWQM.

LEACHM, developed in the USA, is now supported in Australia at Flinders University (John Hutson, personal communication). The LEACHM suite of models has four major models and several utilities. LEACHN describes nitrogen and phosphorus, and LEACHB simulates microbial population dynamics (Hutson, 2000).

McGeachan and Wu (2001) reviewed four European models very thoroughly. They point out that ANIMO, SOILN, DAISY and SUNDIAL all consider the main soil nitrogen dynamic processes but the level of detail varies between the models. In particular, the process of anaerobiosis, the root cause of denitrification, is only modelled in ANIMO. Ammonia volatilization is also only modelled in a limited way in ANIMO and DAISY.

4.3 SIMULATING PROCESSES IN N MODELS

This description follows the outline of the N cycle depicted in Figure 3.1.

1. **Nitrogen fixation:** Additions of manure and fertilizers to soil are inputs to N balance models. Some models simulate urea hydrolysis as a function of soil temperature and aerobic condition. Most, however, assume it to be so rapid that applied urea is treated as NH_4^+ . The challenge here is to sensibly partition organic forms of N into pools of specific stability. More detail on the modelling of carbon pools will be given when mineralization/immobilization is discussed. Several models estimate N fixation by legumes. Some assume the total plant N demand can be met, regardless of the amount of N in the soil, whilst others restrict N fixation based on soil water content, soil NO_3^- and NH_4^+ concentration and plant growth stage (Williams, 1995).
2. **Mineralization and immobilization:** In order to model rates of mineralization and immobilization, it is necessary to define several pools of organic matter, as residues and soil organic matter are very heterogeneous in their chemical and physical compositions. Ma and Shaffer (2001) point out that the main challenge is to decide how to partition organic material among different groups, and even more difficult then is to decide what rate coefficients for breakdown to assign to each pool. They indicate that some models, like RZWQM (Ma *et al.*, 1998) partition pools by making a best judgement of the size of pools and then stabilizing these with known management practices. Some models like DAISY (Jensen *et al.*, 2001) and NCSOIL (Nicolardot *et al.*, 1994) calibrate the pools. Ma and Shaffer (2001) further allude to the fact that although some models give some guidance or default values, like EPIC based on years of cultivation and CENTURY based on residue lignin content, it remains an extremely difficult task to correctly partition the pools, especially considering the dynamic nature of the microbial population.

Ma and Shaffer (2001) describe, in some detail, the differences in approach to modelling organic pools in several models. Most models consider at least one or more surface pools (crop residues and applied manure), and one or more soil organic matter pools. The pools are differentiated from each other in terms of their physical and chemical stability, usually expressed in terms of C:N ratios and decomposition rates or half lives / turn over times.

Ma and Shaffer (2001) point out that the role of microbes in the decomposition process may be explicitly expressed in some models. They further describe the basic approaches to modelling mineralization rate for each organic C pool as follows:

For more complex models Monod kinetics may be used where the decomposition rate (r_i) is described as a function of the size of the microbial biomass (B), the carbon content (C_i) of a specific pool (i), the maximum mineralization rate (μ_m) and a half saturation constant (k_s):

$$r_i = \frac{\mu_m C_i}{k_s + C_i} B$$

If microbial action is not considered, μ_m and B can be combined to give a Michaelis-Menten kinetics equation. In simple models Ma and Shaffer (2001) explain that the mineralization rate equation can be simplified still further to give a first order kinetics equation:

$$r_i = k_1 C_i$$

or a zero-order equation:

$$r_i = k_0$$

The zero-order rate constant k_0 and the first order rate constant k_1 are often modified for soil water content, temperature, pH, O_2 and microbial effects. These simpler approaches assume micro-organisms are not limiting to biological processes, and the zero-order equation further assumes the carbon source to be non-limiting, i.e. the rate is independent of the carbon pool size. The C coming from the decomposition process can be given off as CO_2 , assimilated into microbial biomass or transferred to another carbon pool. The N that is released during this process depends on the C:N ratio of each pool (Molina and Smith, 1998).

Simpler models like NLEAP calculate a net N mineralization rate depending on its C:N ratio and decomposition rate (Hansen *et al.*, 1995). More complex models like RZWQM, on the other hand, calculate both mineralization from the decay of organic pools, and immobilization from the growth of microbes. In this model the C:N ratio does not control the decomposition rate, but rather a first order decay constant modified for soil temperature, O_2 , hydrogen, population of aerobic heterotrophic microbes, aerobic condition and ionic strength (Ma *et al.*, 2001; Shaffer *et al.*, 2000). For immobilization in RZWQM, the *aerobic heterotrophs* grow by decaying the organic pools and net assimilation of N is calculated from the C:N ratios of the microbial biomass and the organic pools. The *autotrophs* grow from energy obtained from nitrification and a fraction of the nitrified N ends up in microbial biomass. *Facultative heterotrophs* use denitrification to grow and an efficiency factor partitions some of the N into microbial biomass. Microbial biomass death is described as a first order kinetics process (Ma and Shaffer, 2001).

3. **Nitrification:** Ma and Shaffer (2001) discuss some of the differences in approaches to simulating nitrification in several models. Just like with mineralization, Monod, first-order and second-order kinetics are used for these calculations, and rates of conversion from NH_4^+ to NO_3^- are modified by water, O_2 , pH and temperature, and for the more sophisticated models, concentrations of autotrophic bacteria and the presence of nitrification inhibitors. Importantly, some of the simpler models do not even simulate nitrification as it is assumed the mineralization is so much slower than nitrification, then it would be reasonable to assume that the first inorganic product of mineralization is NO_3^- and not NH_4^+ . The same assumption can be made for fertilizer NH_4^+ , where it is immediately converted to NO_3^- (Hansen *et al.*, 1995). Some models also estimate N_2O production from nitrification dependent on N turnover rate and excess NH_4^+ in the soil (Parton *et al.*, 1996; Shaffer *et al.*, 2001).
4. **Plant uptake:** This is a crucial part of all N balance models and relates to plant growth, root distribution, C and N dynamics, water movement and N transport (Ma & Shaffer, 2001). N uptake is described in most models as a process either limited by supply or demand. Demand for plant N depends on crop growth and optimal N concentration, but luxury nutrition is simulated in some models. The CERES model (Godwin and Jones, 1991) describe N demand as the N deficiency of existing biomass plus N required for new growth.

As far as N supply to the plant is concerned, this is determined by plant available total N (NH_4^+ and NO_3^-) and rate of water uptake, so called passive uptake. In some models, if passive uptake cannot meet plant demand, an active uptake of N is initiated to attempt to meet plant demand (Hanson, 2000; Hutson, 2000). Some models distinguish between NH_4^+ and NO_3^- uptake, but many do not.

5. **Leaching:** NO_3^- leaching is mechanistically simulated in models with detailed water and solute flux descriptions. The amount of N leached will depend on the downward flux of water below the root zone, and its nitrate concentration.

In the somewhat simpler NLEAP (Shaffer *et al.*, 1991), NO_3^- leaching is computed using an exponential relationship based on water available for leaching and soil porosity. In the monthly time step CENTURY model (Parton *et al.*, 1993), nitrate leaching is a function of monthly-saturated water flow and percentage sand.

6. **Denitrification:** Ma and Shaffer (2001) allude to the fact that denitrification simulations are extremely empirical, not only due to the poor understanding of the process itself, but also because of the spatial and temporal variability of anaerobic soil conditions. Once again we have zero or first order kinetics, or a Michaelis-Menten approach.

Some models do not simulate denitrification, some just assume a fraction of the N is lost in the gaseous form. The more sophisticated models distinguish between N_2 and N_2O loss and take into account soil anaerobic conditions, temperature, pH, soil carbon substrate and the population of denitrifying bacteria.

Several models estimate denitrification once the soil water content exceeds a certain threshold (Williams, 1995; Godwin and Singh, 1998).

7. **Ammonia volatilization:** Ma and Shaffer (2001) point out that considerable quantities of NH_3 may be lost when there is a high concentration of NH_4^+ close to the soil surface, especially under conditions of high pH. Most models that simulate volatilization use a first-order kinetics approach influenced by temperature. Other important factors considered in NLEAP are fertilizer application method, occurrence of precipitation, CEC and residue cover (Shaffer *et al.*, 1991; Hansen *et al.*, 1995). In the more sophisticated RZWQM, the rate coefficient depends on the pressure gradient of NH_3 between the soil and air, temperature and wind speed (Ma *et al.*, 2001).

Several models do not bother to simulate ammonia volatilization at all, and other simple approaches are to assume a fixed fraction of N mineralised is volatilised. Interestingly, EPIC (Williams, 1995) assumes NH_3 can be volatilised from the whole soil profile, and not just the uppermost soil layer. In GLEAMS (Knisel, 1993), ammonia volatilization is only simulated for surface applied solid, slurry and animal waste.

8. **Ammonia Fixation:** Judging from the comprehensive review on nine nutrient balance models from the USA, ammonia fixation is not generally simulated. The more detailed models that include chemical equilibria routines and cation exchange, however, will most likely include NH_4^+ as one of the cations in solution, and will therefore estimate the interactions this cation will undergo with the solid phase.

There are big differences between models, but as Ma & Shaffer (2001) point out, there are no right or wrong approaches. Models are unique, and each has its own strong and weak points. Model users need to be supplied with support and detailed guidelines on how to parameterise the models.

A real problem with C:N models is that there are no experimental methods to adequately differentiate the somewhat arbitrarily defined organic pools (Ma and Shaffer, 2001). As a result of not examining the physical and chemical properties of the pools, parameters associated with them remain empirical or unknown. For these reasons multiple pool models may not have significant advantages in terms of solving "real-world" problems (Ma and Shaffer, 2001).

Ma and Shaffer (2001) feel that the future direction of model development should be to adopt a modular modelling framework, where users can assemble their own systems models from well-defined modules for their specific applications. This sentiment is fortunately echoed by other model developers and the Australian APSIM approach bears testimony to this.

4.4 PHOSPHORUS MODELLING

Jones *et al.* (1991) point out that although considerable information is available on the complex chemical and physiological transformations of P in the soil, and its uptake by plants, attempts to formulate comprehensive, management-oriented simulation models of P cycling have been limited. Models on P cycling range from complex mathematical representations of soil P pools and plant uptake, to relatively simple predictors of fertilizer P requirements for crops.

On a coarser scale, a development version of ACRU that considers P loading at a catchment scale has been used in the Umgeni Catchment (Kienzie *et al.*, 1997)

4.5 SIMULATING PROCESSES IN P MODELS

Some of the earliest work was done by Nye and Marriott (1969), when they described the movement of P to individual roots by mass flow and diffusion. Absorption kinetics of the root is assumed to follow Michaelis-Menten (M-M) kinetics, after subtracting a term for efflux (Jones *et al.*, 1991). These authors point out that the Nye and Marriott uptake model requires as soil parameters; an effective average diffusion coefficient, initial P concentration in solution and buffer power, whilst plant parameters required are; maximum influx rate, the M-M constant, water influx, root radius, initial root length and rate of root growth. Jones *et al.* (1991) feel that although this early model did not describe the cycling of P in organic and inorganic pools, it did quantify the importance of several factors on P uptake by roots. These were, amongst others, the effect of soil temperature, root morphology and fertilizer placement.

Jones *et al.* (1991) describe the P transformation and transport under steady flow conditions model of Mansell *et al.* (1977). Five phases of inorganic P were considered: solution, physically adsorbed, chemisorbed, immobilized and precipitated. First-order kinetics is used to describe flow among the phases. For practical applications, methods are needed to assign rate constants based on soil properties, and to simulate organic P transformations and their effects on inorganic pools.

5. MITIGATION MEASURES / BEST MANAGEMENT PRACTICES

5.1 MANAGING NITROGEN AND PHOSPHORUS FOR MAXIMUM PRODUCTIVITY AND MINIMUM POLLUTION

Non-point source pollution by N and P from agricultural systems, because of their dispersed nature, has been and will always be, difficult to quantify and control. A comprehensive consideration of sources and control of non-point N and P pollution from agricultural systems is beyond the scope of this section. However, a concise discussion on some aspects in this regard follows.

In South Africa 85% of the agricultural land is used for livestock production, while the remaining 15% of agricultural land is used for crop production. Generally, crop production is regarded as the major contributor to non-point source pollution. However, livestock production cannot be neglected. When livestock production results in degradation of veld it has a large influence on the hydrological cycle and also the C, N and P cycles. Usually, veld degradation is associated with less vegetative cover with lower biomass production. As a result of this, soil organic matter declines, with negative affects on soil aggregation, and hence more wind and water erosion. With this accelerated mineralization process of soil organic matter, excessive N and P are released which contribute to pollution. Good management practices to maintain good veld condition are of utmost importance in South Africa due to the large portion of agricultural land under livestock production.

Usually, it is assumed that non-point source pollution from either N or P increases when crop production intensifies. However, the sources and control of non-point N and P pollution are more or less the same regardless of the intensity of crop production and will be discussed simultaneously.

Designing efficient management programs for either N or P in cropping systems to sustain maximum productivity with minimum environmental impact, requires not only knowledge of the two cycles, but also of the soils, crops and fertilizers involved. These interactions are also influenced by, inter alia, climatic conditions, topographic position and cultivation practices.

It would be fortunate indeed if a relatively simple set of data for a catchment could be fed into a computer that would then print out a general recommendation for farms in such a catchment. However, there are great differences among farms in a catchment and even fields on a farm with respect to kinds of soils, crops and fertilizers, for example. Efficient N and P management programs, therefore, need to be derived individually for each field or set of fields of similar nature that may be called an ecotope.

By far the best way to minimize N and P non-point source pollution from cropping systems, is to focus on practices that favour high crop yields and reduce soil erosion. Some of these practices are listed below:

- Concentrate crops in suitable geographic regions, viz. where the climate and soils are most favourable for production.
- Select the best available genetic hybrids or crop varieties that are suitable for a specific geographic region.
- Plant at the optimum time and density that permits the crop to most advantageously utilize available solar energy.
- Introduce the best cropping sequence for the most efficient utilization of applied N and P.
- Use appropriate soil conservation practices like terracing, contouring, strip cropping, crop residues or vegetative cover to combat either wind or water erosion.

- Improve water management in order to limit periods of either excessive or deficient water in soil.
- Plant cover crops where possible to reduce soil erosion and enhance nutrient uptake.
- Control weeds, diseases and insects efficiently for optimum crop growth and development
- Adapt tillage practices to ensure good root development for optimum water and nutrient uptake.

Usually, for a given situation, a specific combination of the abovementioned practices will be most appropriate, and will constitute best management practise.

In addition to the points listed, which are by no means comprehensive, a sound fertilization program is essential for sustainable crop production. A sound fertilization program should take care firstly of soil pH, and secondly of the availability of essential plant nutrients. The availability of most plant nutrients is affected to some or other extent by pH.

Only the approach towards sound N and P fertilization programs will be discussed here. The broad general goal should be to make the best possible match between the amount of N and P needed by a crop and the amount available from all sources during the growing season. This goal can be achieved through a series of sequential steps, some of which are relatively complex. These major steps are more or less similar for N and P, and are followed to a large extent by South African commercial farmers to limit either over or under fertilization. However, for several reasons subsistence farmers do not necessarily follow these steps.

The steps are as follows:

1. Estimate the probable crop yield based upon history or modelling. Crop yield is a determining factor in the amount of N or P to be applied.
2. Assess the availability of N and P in soil. The approach with regard to N and P differs from one another.

Nitrogen: This includes the N inherent in the soil organic matter, the amount from residues of preceding crops, and the carry-over from previous applications. One may expect that agriculture would be sufficiently sophisticated to precisely test the soil for N that will become available within a growing season, but research on the subject over the past 50 years has had only limited success.

Phosphorus: Much more success has been achieved in this regard with soil testing for critical extractable soil P concentrations. Critical extractable soil P concentrations were established above which little or no crop response to P fertilization is obtained. These critical values vary with the extractant used and between soils of different texture.

3. Consider the N and P that will be applied from other outside sources if any.
4. Decide upon the amount of fertilizer N and P needed for optimum economic return, modified to the extent feasible through environmental considerations.
5. Select best time of application. Some differences exist in this regard between N and P.

Nitrogen: The fundamental principle is to supply N as close as possible to the time it is needed by the crops as a result of its high mobility. This approach integrates at least four factors, viz. the crop to be grown, climate during growing season, characteristics of the soil and properties of the fertilizer. Usually, only a portion of the required N is applied at planting and the remaining N is either side- or top-dressed during the growing season to increase efficiency and decrease the risk of pollution.

Phosphorus: As a result of the immobility of P in most soils, it is applied just before or at planting. This reduces the farming risk of the conversion of water soluble P to less available forms and thereby increases the efficiency of the applied P. However, if water insoluble P is to be used, it should be applied well in advance of planting to allow enough time for dissolution.

6. Choose best method of application. Determining the proper zone in the soil to apply nutrients, is just as important as choosing the correct amount to apply. Numerous placement methods have been developed, and the following factors should be considered with fertilizer placement decisions, viz. efficient use of nutrients from plant emergence to maturity, prevention of salt injury to seedlings, and convenience to the farmer. Fertilizer placement options generally involve broadcast, subsurface band and surface band before planting, subsurface band, seed band and surface band at planting, and side-dressing and topdressing after planting. The selection of a suitable placement practice depends on crop characteristics, soil properties, soil test level, nutrient mobility and equipment availability. In South Africa, especially for dryland crop production, both N and P are usually subsurface banded with either side- or topdressing of N during the growing season to improve their efficiency.

For more information on managing N and P for maximum productivity and minimum pollution, readers are directed to the reviews of Keeney (1982), Aldrich (1984), Sharpley *et al.* (1999) and Havlin *et al.* (1999).

A recent innovation that shows promise in reducing surface and subsurface flow of N and P from arable land is riparian buffer zones. Some results obtained with riparian buffer zones are presented in Table 5.1. The efficiencies of riparian buffer zones for N and P removal have yet to be accurately established, and costs and novelty will limit adoption. The clear benefit of nutrient removal by trees will have to be weighed up against their increased water use over that of the natural vegetation in a water scarce country like South Africa. Many exotic tree species are actively being removed from riparian zones in the "Working for Water" campaign initiated by the Department of Water Affairs and Forestry. In fact, forestry has been declared an SFRA (Stream Flow Reduction Activity).

Table 5.1 Effects of different size riparian buffers on reductions of sediment and nutrients from field surface runoff (Havlin *et al.*, 1999)

Buffer width (m)	Buffer type	Input concentration (mg l ⁻¹)	Output concentration (mg l ⁻¹)	Reduction ^a %
Sediments				
4.6 ^c	Grass	7284	2841	61.0
9.2 ^c	Grass	7284	1852	74.6
19.9 ^d	Forest	6480	661	89.8
23.6 ^e	Grass/forest	7284	290	96.0
28.2 ^f	Grass/forest	7284	168	97.4
Nitrogen^b				
4.6 ^c	Grass	14.11	13.55	4.0
9.2 ^c	Grass	14.11	10.91	22.7
19.9 ^d	Forest	27.59	7.08	74.3
23.6 ^e	Grass/forest	14.11	3.48	75.3
28.2 ^f	Grass/forest	14.11	2.80	80.1
Phosphorus^b				
4.6 ^c	Grass	11.30	8.09	28.5
9.2 ^c	Grass	11.30	8.56	24.2
19.9 ^d	Forest	5.03	1.51	70.0
23.6 ^e	Grass/forest	11.30	2.43	78.5
28.2 ^f	Grass/forest	11.30	2.57	77.2

^a Percentage of reduction = 100(input-output)/input.

^b Nitrogen = Nitrate-N + exchangeable particle ammonium + dissolved ammonium + particle organic N + dissolved organic N. Phosphorus = particle P + dissolved P.

^c Calculated from masses of total suspended solids, total N, total P, runoff depth and plot size (22 x 5m)

^d Surface runoff concentrations at 19 m into forest. N and P constituents same as input.

^e 4.6 m grass buffer plus 19 m of forest.

^f 9.2 m grass buffer plus 19 m of forest.

6. SUMMARY AND RESEARCH NEEDS

It is clear that agriculture contributes to non point source pollution of ground and surface waters. It is not, however, a trivial exercise to determine exactly where and to what extent agriculture contributes to this problem. It is essential, however, that problem areas are identified, and the extent of the problem quantified, so that appropriate steps can be taken to reverse, halt or minimize the impact on the environment.

A great deal is known about the N and P cycles, and great strides have been made in modelling these interactions. However, many processes have to be empirically described, as purely mechanistic approaches are either too poorly understood, or it is too difficult to parameterise such models. There is a general lack of expertise locally in modelling N and P balances, and there is a clear need to integrate nutrient balance models with locally used water balance models. Such models will be invaluable in assessing the long-term impact of dry land and irrigated cropping systems on nitrate leaching to groundwater, and will help to identify potential problem areas. It would also be desirable to integrate the output from field nutrient and water balance models, possibly run on a catchment scale, with groundwater simulations, to assess the impact of different cropping systems or land uses on groundwater reserves. Such modelling tools would also be extremely useful for designing best management practices for diverse situations, and could be used to design robust waste disposal and effluent irrigation schemes.

Rough field scale nutrient balances suggest that dry land cropping in South Africa has a limited impact on groundwater nitrate levels, especially on deeper soils. However, the irrigation and feedlot sectors of the agricultural economy should be more closely scrutinized. Most of the nutrients applied to crops and removed in the harvested products are concentrated in urban areas and feedlots by the consumers of these products. Water care works and intensive animal feeding units, therefore, are large sources of pollution, and better management of these facilities deserves more attention.

Tredoux (1993) qualitatively identified the major contributors to groundwater N pollution. This now needs to be done more quantitatively. Areas and aquifers at risk need to be delimited, and the major contributors or potential contributors to non-point source pollution need to be identified. Detailed studies, as recommended by Tredoux (1993), could and should be undertaken at selected vulnerable sites to better identify and understand the processes involved.

Bond (1998), feels that guidelines for managing waste are at present very general, and they can be made much more rigorous by improving our understanding and ability to predict soil processes. He also feels that there have been advances in the availability and usability of reliable numerical models of soil water dynamics, like HYDRUS (Kool and Van Genuchten, 1991), LEACHM (Hutson and Wagenet, 1995) and SWIM (Ross, 1990; Verburg *et al.*, 1996). He does not, however, feel that the quantitative prediction of the fate of N, particularly through denitrification and volatilization, is that reliable, and recommends further work on this aspect.

Bond (1998) sees an opportunity to, through better understanding of the processes involved, increase gaseous losses of N under effluent irrigation, but he cautions that some of the gases produced are deemed to be "greenhouse gases". Bond (1998) also challenges the soil science community to develop effective, inexpensive and assessable measurement techniques, and the establishment of performance criteria that indicate when management changes are required.

7. REFERENCES

- Ahuja, L.R., Rojas, K.W., Hanson, J.D., Shaffer, M.J. and Ma, L. (Eds.), 2000. Root zone water quality model: Modeling management effects on water quality and crop production. Water Resources Publications, LLC, Highlands Ranch, CO. p 372.
- Aldrich, S.R., 1984. Nitrogen management to minimize adverse effects on the environment. In R.D. Hauck (Ed.). Nitrogen in crop production. Am. Soc. of Agron., Madison, Wisconsin.
- Baard, J.A.J., 1992. Written communication. Elsenburg Agricultural Development Institute, Elsenburg.
- Barrow, N.J., 1985. Reactions of anions and cations with variable charge soils. *Adv. Agron.* 38, 183 – 230.
- Berghuijs van Dijk, J.T., Rijtema, P.E. and Roest, C.W.J., 1985. ANIMO agricultural nitrogen model. Nota 1671, Institute for Land and Water Management Research, Wageningen, The Netherlands.
- Bierman, C.R., 2001. Grondvrugbaarheid en bemesting in die Noordwesprovinsie met verwysing na volhoubare droëlandgewasverbouing. Magister in Volhoubare Landbou Skripsie, Universiteit van die Vrystaat, Bloemfontein.
- Biesenbach, F.W., 1984. 'n NPK-balansstaat vir die landbougrond van die Republiek van Suid-Afrika. *Tegniese Mededeling* 187, 60 – 91. Departement Landbou, Pretoria.
- Bond, W.J., 1998. Effluent irrigation – an environmental challenge for soil science. *Aus. J. Soil Res.*, 36, 543-55.
- Bradbury, N.J., Whitmore, A.P., Hart, P.B.S. and Jenkinson, D.S., 1993. Modelling the fate of nitrogen in crops and soils in the years following application of ^{15}N -labelled fertilizer to winter wheat. *Journal of Agricultural Science, Cambridge*, 121: 363 – 379.
- Cameron, K.C. and Haynes, R.J., 1986. Retention and movement of nitrogen in soils. In R.J. Haynes (Ed.). *Mineral nitrogen in the plant-soil system*. Academic Press, New York.
- Casey, N.H. and Meyer, J.A., 1996. Interim water quality guidelines for livestock watering. WRC Report TT76/96.
- Colvin, C., 2000. Handbook of groundwater quality protection for farmers. WRC Report No. TT 116/00, Pretoria.
- Department of Water Affairs and Forestry, South Africa. 1996. South African Water Quality Guidelines (2nd ed) Volume 4: Agricultural Use: Livestock Watering.
- Department of Water Affairs, Namibia (DWA-Namibia). 1977. Watergehalte in Suidwes-Afrika. Dept Water Affairs, Windhoek, p31.
- Du Preez, C.C. and Du Toit, M.E. 1995. Effect of cultivation on the nitrogen fertility of selected agro-ecosystems in South Africa. *Fertilizer Research* 42, 27 – 32.
- Du Toit, M.E., Du Preez, C.C., Hensley, M. and Bennie, A.T.P. 1994. Effek van bewerking op die organiese materiaalinhoud van geselekteerde droëlandgronde in Suid-Afrika. *S.Afr. Tydskr. Plant Grond* 11, 71 – 79.
- Firestone, M.K. 1982. Biological denitrification. In F.J. Stevenson (Ed.). *Nitrogen in agricultural soils*. Am. Soc. of Agron., Madison, Wisconsin.
- Franco, U., Oelschlägel, B. and Schenk, S. 1995. Simulation of temperature, water and nitrogen dynamics using the model CANDY. *Ecological Modelling*, 81: 213 – 222.

- Frossard, E., Brossard, M., Hedley, M.J. and Metherall, A. 1995. Reactions controlling the cycling of phosphorus in soils. In H. Tiessen (Ed.). *Phosphorus in the global environment: transfers, cycles and management*. John Wiley and Sons, New York.
- Garnier, M., Lo Porto, A., Marini, R. and Leone, A. 1998. Integrated use of GLEAMS and GIS to prevent groundwater pollution caused by agricultural disposal of animal waste. *Environmental Management* 22, 747 – 756.
- Godwin, D.C. and Jones, C.A. 1991. Nitrogen dynamics in soil-plant systems. In J. Hanks and J.T. Ritchie (Eds.). *Modeling plant and soil systems*. American Society of Agronomy, Inc. Madison, WI, 287 – 321.
- Godwin, D.C. and Singh, U. 1998. Nitrogen balance and crop response to nitrogen in upland and lowland cropping systems. In G.Y. Tsuji, G. Hoogenboom and P.K. Thornton (Eds.). *Understanding options for agricultural production*. Kluwer Academic, Boston, 55 – 77.
- Grant, R.F., 2001. Modeling transformations of soil organic carbon and nitrogen at differing scales of complexity. In M.J. Shaffer, L. Ma, and S. Hansen (Eds.). *Modeling carbon and nitrogen dynamics for soil management*. Lewis Publishers, Boca Raton, FL, 599 – 632.
- Hanks, J. and Ritchie, J.T., 1991. *Modeling plant and soil systems*. American Society of Agronomy, Inc., Madison, WI, pp 545.
- Hansen, S., Jensen, H.E., Nielsen, N.E. and Svendsen, H. 1991. Simulation of nitrogen dynamics and biomass production in winter wheat using the Danish simulation model DAISY. *Fertilizer Research* 27, 245 – 259.
- Hansen, S., Jensen, H.E. and Shaffer, M.J., 1995. Development in modeling nitrogen transformations in soil. In P.E. Bacon (Ed.). *Nitrogen fertilization in the environment*. Marcel Dekker, New York, 83 – 107.
- Hanson, J.D. 2000. Generic Crop Production. In L.R. Ahuja, K.W. Rojas, J.D. Hanson, M.J. Shaffer and L. Ma. (Eds.). *Root Zone Water Quality Model: Modeling management effects on water quality and crop production*. Water Resources Publications, LLC, Highlands Ranch, CO, 81 – 118.
- Havelka, U.D., Boyle, M.G. and Hardy, R.W.F. 1982. Biological nitrogen fixation. In F.J. Stevenson (Ed.). *Nitrogen in agricultural soils*. Am. Soc. of Agron., Madison, Wisconsin.
- Havlin, J.L., Beaton, J.D., Tisdale, S.L. and Nelson, W.L., 1999. *Soil fertility and fertilizers: an introduction to nutrient management*. Prentice Hall, New Jersey.
- Haynes, R.J. 1986a. Origin, distribution and cycling of nitrogen in terrestrial ecosystems. In R.J. Haynes (Ed.). *Mineral nitrogen in the plant-soil system*. Academic Press, New York.
- Haynes, R.J. 1986b. The decomposition process: mineralization, immobilization, humus formation and degradation. In R.J. Haynes (Ed.). *Mineral nitrogen in the plant-soil system*. Academic Press, New York.
- Haynes, R.J. 1986c. Nitrification. In R.J. Haynes (Ed.). *Mineral nitrogen in the plant-soil system*. Academic Press, New York.
- Haynes, R.J. 1986d. Uptake and assimilation of mineral nitrogen by plants. In R.J. Haynes (Ed.). *Mineral nitrogen in the plant-soil system*. Academic Press, New York.
- Haynes, R.J. and Sherlock, R.R. 1986. Gaseous losses of nitrogen. In R.J. Haynes (Ed.). *Mineral nitrogen in the plant-soil system*. Academic Press, New York.
- Hutson, J.L. 2000. *LEACHM: Model Description and User's Guide*. School of Chemistry Physics, and Earth Sciences, The Flinders University of South Australia, Adelaide, South Australia.

- Hutson, J.L. and Wagenet, R.J. 1995. An overview of LEACHM: A process based model of water and solute movement, transformations, plant uptake and chemical reactions in the unsaturated zone. In R.H. Loeppert, A.P. Schwab and S. Goldberg (Eds.). *Chemical equilibrium and reaction in soils*, pp 409 – 22. SSSA Special Publication No 42 (Soil Science Society of America: Madison, WI).
- Jansson, S.L. and Persson, J. 1982. Mineralization and immobilization of soil nitrogen. In F.J. Stevenson (Ed.). *Nitrogen in agricultural soils*. Am. Soc. of Agron., Madison, Wisconsin.
- Jensen, L.S., Mueller, R., Bruun, S and Hansen, S. 2001. Application of the DAISY model for short and long-term simulation of soil carbon and nitrogen dynamics. In M.J. Shaffer, L. Ma, and S. Hansen (Eds.). *Modeling carbon and nitrogen dynamics for soil management*. Lewis Publishers, Boca Raton, FL, 485 – 511.
- Johnson, H.L., Bergström, P.-E. Janson, and Paustian, K. 1987. Simulated nitrogen dynamics and losses in a layered agricultural soil. *Agriculture, Ecosystems and Environment*, 18: 333-356.
- Jones, C.A., Bland, J.T. Ritchie, Williams, J.R. 1991. Simulation of root growth. In J. Hanks and J.T. Ritchie (Eds.). *Modeling plant and soil systems*. American Society of Agronomy, Inc., Madison, WI, 91 – 123.
- Keeney, D.R. 1982. Nitrogen management for maximum efficiency and minimum pollution. In F.J. Stevenson (Ed.). *Nitrogen in agricultural soils*. Am. Soc. of Agron., Madison, Wisconsin.
- Kienzie, S.W., Lorentz, S.A. and Schulze, R.E. 1997. Hydrology and water quality management of the Mgeni catchment. WRC Report No. TT87/97, p88.
- Knisel, W.G. (Ed.). 1993. GLEAMS, Groundwater loading effects of agricultural management systems. Version 2.10. UGA-CPES-BAED, publication No. 5. 259 pp.
- Kool, J.B. and Van Genuchten, M. 1991. HYDRUS: One-dimensional variably saturated flow and transport model, including hysteresis and root water uptake. (US Salinity Laboratory, Riverside, CA).
- Leonard, R.A., Knisel, W.G. and Still, D.S. 1987. GLEAMS: groundwater loading effect of agricultural management systems. *Trans. ASAE*, 30: 1403 – 1418.
- Li, C., Frolking, S. and Frolking, T.A. 1992. A model of nitrous oxide evolution from soil driven by rainfall events. 1. Model structure and sensitivity. *J. Geophys. Res.*, 97: 9777 – 9796.
- Lobe, I., Amelung, W. and Du Preez, C.C. 2001. Losses of carbon and nitrogen with prolonged arable cropping from sandy soils of the South African Highveld. *Eur. J. Soil Sci.* 52, 13 – 101.
- Ma, L. and Shaffer, M.J. 2001. A review of carbon and nitrogen processes in nine U.S. soil nitrogen dynamics models. In M.J. Shaffer, L. Ma, and S. Hansen (Eds.). *Modeling carbon and nitrogen dynamics for soil management*. Lewis Publishers, Boca Raton, FL, 55 – 102.
- Ma, L., Shaffer, M.J., Boyd, J.K., Waskom, R., Ahuja, L.R., Rojas, K.W. and Xu, C. 1998. Manure management in an irrigated silage corn field: experiment and modeling. *Soil Sci. Soc. Am. J.*, 62, 1006 – 1017.
- Ma, L., Shaffer, M.J. and Ahuja, L.R. 2001. Application of RZWQM for soil nitrogen management. In M.J. Shaffer, L. Ma, and S. Hansen (Eds.). *Modeling carbon and nitrogen dynamics for soil management*. Lewis Publishers, Boca Raton, FL, 265 – 301.
- McGechan, M.B. and Wu, L. 2001. A Review of carbon and nitrogen processes in European soil nitrogen dynamics models. In M.J. Shaffer, L. Ma, and S. Hansen (Eds.). *Modeling carbon and nitrogen dynamics for soil management*. Lewis Publishers, Boca Raton, FL, 103 – 171.
- Molina, J.A.E., Clapp, C.E., Shaffer, M.J., Chichester, F.W. and Larson, W.E. 1983. NCSOIL, A model of nitrogen and carbon transformation in Soil: Description, calibration, and behaviour. *Soil Sci. Soc. Am. J.*, 47: 85 – 91.

- Molina, J.A.E. and Smith, P. 1998. Modeling carbon and nitrogen processes in soils. *Adv. Agron.*, 62: 253 – 298.
- Nicolardot, B., Molina, J.A.E. and Allard, M.R. 1994. C and N fluxes between pools of soil organic matter. Model calibration with long-term incubation data. *Soil Biology and Biochemistry* 26: 234 – 243.
- Nielsen, D.R., Biggar, J.W. and Wierenga, P.J., 1982. Nitrogen transport processes in soil. In F.J. Stevenson (Ed.). *Nitrogen in agricultural soils*. Am. Soc. of Agron., Madison, Wisconsin.
- Nommik, H. and Vahtras, K. 1982. Retention and fixation of ammonium and ammonia in soils. In F.J. Stevenson (Ed.). *Nitrogen in agricultural soils*. Am. Soc. of Agron., Madison, Wisconsin.
- Nye, P.H. and Marriott, F.H.C. 1969. A theoretical study of the distribution of substances around roots resulting from simultaneous diffusion and mass-flow. *Plant Soil* 30: 459 – 472.
- Olsen, R.A. and Kurtz, L.T. 1982. Crop nitrogen requirements, utilization and fertilization. In F.J. Stevenson (Ed.). *Nitrogen in agricultural soils*. Am. Soc. of Agron., Madison, Wisconsin.
- Parton, W.J., Mosier, A.R., Ojima, D.S., Valentine, D.W., Schimel, D.S., Weier, K. and Kulmala, A.E. 1996. Generalized model for N_2 and N_2O production from nitrification and denitrification. *Global Biogeochemical Cycles*, 10: 401 – 412.
- Parton, W.J. and Rasmussen, P.E. 1994. Long-term effects of crop management in wheat/fallow: II. CENTURY model simulations. *Soil Sci. Soc. Am. J.*, 58, 530 – 536.
- Parton, W.J., Scurlock, J.M.O., Ojima, D.S., Gilmanov, T.G., Scholes, R.J., Schimel, D.S., Kirchner, T., Menaut, J.-C., Seastedt, T., Garcia Moya, E., Kamnalrut, A. and Kinyamario, J.I. 1993. Observations and modelling of biomass and soil organic matter dynamics for the grassland biome worldwide. *Global Biogeochemical Cycles*, 7: 785 – 809.
- Ross, P.J. 1990. SWIM – a Simulation Model for Soil Water Infiltration and Movement: Reference Manual. CSIRO Division of Soils, Adelaide.
- Schmidt, E.L. 1982. Nitrification in soil. In F.J. Stevenson (Ed.). *Nitrogen in agricultural soils*. Am. Soc. of Agron., Madison, Wisconsin.
- Shaffer, M.J., Halvorson, A.D. and Pierce, F.J., 1991. Nitrate leaching and economic analysis package (NLEAP): Model description and application. In R.F. Follet, et al. (Eds.). *Managing nitrogen for groundwater quality and farm profitability*. Soil Sci. Soc. of America, Madison, WI, 13, 285 – 322.
- Shaffer, M.J. and Larson, W.E. (Eds.). 1987. NTRM, a soil-crop simulation model for nitrogen, tillage, and crop-residue management. U.S. Department of Agriculture Conservation Research Report No. 34-1, pp 103.
- Shaffer, M.J., Ma, L. and Hansen, S., 2001. Modeling carbon and nitrogen dynamics for soil management. Lewis Publishers, Boca Raton.
- Shaffer, M.J., Rojas, K.W., DeCoursey, D.G. and Hebson, J.D. 2000. Nutrient chemistry processes – OMNI. In L.R. Ahuja, K.W. Rojas, J.D. Hanson, M.J. Shaffer, and L.Ma (Eds.). *Root zone water quality model, modeling management effects on water quality and crop production*. Water Resources Publications, Highlands Ranch, CO, 119 – 144.
- Shaffer, M.J., Schumacher, T.E. and Ego, C.L. 1995. Simulating the effects of erosion on corn productivity. *Soil Sci. Soc. Am. J.*, 59: 672 – 676.
- Sharpley, A.M., Sims, J.T. and Pierzynski, C.M. 1994. Innovative soil phosphorus indices: assessing inorganic phosphorus. In J.L. Havlin (Ed.). *New directions in soil testing for nitrogen, phosphorus and potassium*. Soil Sci. Soc. of Am., Madison, Wisconsin.

- Simonis, M. 1999. The assessment of ambient groundwater quality, at a national scale in the Republic of South Africa. Final report to the Water Research Commission, WRC project K5/841, Pretoria.
- Stevenson, F.J. 1982. Origin and distribution of nitrogen in soil. In F.J. Stevenson (Ed.). Nitrogen in agricultural soils. Am. Soc. Agron., Madison, Wisconsin.
- Stevenson, F.J. and Cole, M.A. 1999. Cycles of soils: carbon, nitrogen, phosphorus, sulphur, micronutrients, 2nd edn. John Wiley & Sons, New York.
- Syers, J.K. and Curtin, D. 1989. Inorganic reactions controlling phosphorus cycling. In H. Tieszen (Ed.). Phosphorus cycles in terrestrial and aquatic ecosystems. Saskatchewan Institute of Pedology, Saskatchewan, Canada.
- Tredoux, G. 1993. A preliminary investigation of the nitrate content of groundwater and limitation of the nitrate input. WRC Report No. 368/1/93, Pretoria.
- Van Ginkel, C. 2002. Trophic status assessment – Executive Summary. Institute for Water Quality Studies, Department of Water Affairs and Forestry, Pretoria.
- Verburg, K., Ross P.J. and Bristow, K.L. 1996. SWIM Version 2 Users Manual. CSIRO Division of Soils, Divisional Report No. 130.
- Wamsley, R.D. 2000. Perspectives on eutrophication of surface waters: Policy/Research needs in South Africa. WRC Report No. KV129/00, Pretoria.
- Williams, J.R. 1995. The EPIC model. In V.P. Singh (Ed.). Computer models of watershed hydrology. Water Resources Publications, Highlands Ranch, CO, 909 – 1000.
- World Health Organization (WHO). 1985. Health hazards from nitrates in drinking water. Copenhagen, p 102.
- Young, C.P., Oakes, D.B. and Wilkonson, W.B. 1976. Prediction of future nitrate concentrations in groundwater. Groundwater 14, 426 – 438.

DISCUSSION PAPER 2:

PESTICIDES – OVERVIEW OF THE EXTENT OF THE PROBLEM AND MODELLING

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MODELLING NON-POINT SOURCE POLLUTION IN AGRICULTURE FROM FIELD TO CATCHMENTS SCALE – A SCOPING EXERCISE - PESTICIDES

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2. OBJECTIVES

The project objectives of this study are listed below:

1. To produce first order estimates of the contribution that agricultural activities are making towards non-point source (NPS) pollution. (The sources of water pollution including fertiliser leaching and wash-off from human settlements, sediments as a result of erosion, animal production units, and the application of insecticides, fungicides and herbicides, etc, are grouped into diffuse or non-point source (NPS) polluters);
2. To assess the level of understanding of the processes operating at various scales for main sources of agricultural NPS pollution - from microscopic level to catchment scale;
3. To assess the range of mitigating measures that are available to address each of the major sources of agricultural NPS pollution; and
4. To assess the predictive (modelling) ability that is available to address each of the major sources of agricultural NPS pollution.

3. GENERAL INTRODUCTION

The climate in South Africa is characterised by warm and dry summers, with winter temperatures falling below 0 °C in inland areas. The summer temperature maxima frequently rises above 35°C. South Africa's most limiting natural resource is water. The average annual rainfall is approximately 500mm, which is considerably less than the world average of 860mm. Most of the central and eastern parts of the country enjoy summer rainfall, whilst the winter rainfall region occurs along the western and south-western side of the country. In order to supply water to industry, agriculture, and domestic users, most of the major rivers have been dammed or have water abstraction schemes in place. Population growth, increased economic activity and changes in land use have all led to increased water demand, which may potentially lead to increased pollution levels.

The water requirements of different sectors in South Africa are shown in Table 3.1 (Basson *et al.*, 1997). These statistics show that the largest users are the irrigation and afforestation (62%) sectors. Even with the projected use in 2030 of 52%, this sector is likely to remain the major user.

Table 3.1 Summary of water requirements per sector for 1996 and 2030 (Basson *et al.*, 1997)

Water Sector	Estimate 1996		Predicted 2030		% Increase from 1996 to 2030
million m ³ per annum	Relative Contribution		million m ³ per annum		Relative Contribution
Urban and Domestic	2171	10.8%	6936	22.8%	219.5%
Mining and Industrial	1598	8.0%	3380	11.1%	111.5%
Irrigation and Afforestation	12344	61.6%	15874	52.2%	28.6%
Environmental	3932	19.6%	4225	13.9%	7.5%
TOTAL	20045	100%	30415	100%	51.7%

Similarly, groundwater demand has shown that irrigation requires approximately 78%, followed by rural domestic 7%, urban 6%, stock watering 6% and mining and quarries 5% (Department of Water Affairs and Forestry).

Agriculture not only plays an important role as a water user, but also in South African water quality. One of the potentially negative impacts from agricultural activities has been identified as the movement of agricultural chemicals into rivers and the environment. Degradation of water quality, specifically with respect to salinity, eutrophication (nutrient enrichment), sediments, pathogens, pesticides and some heavy metals have been attributed to agriculture.

The contribution that irrigation activities make towards the salinisation of water resources has been the main focus of many completed and ongoing studies in South Africa. Although other agricultural activities that give rise to water pollution have been investigated, the present level of knowledge concerning these activities is not as advanced. A strong motivation to study these potential pollution sources therefore exists.

Aspects that require investigation include an assessment of the contribution of the different agricultural activities to the different manifestations of pollution. It will also be necessary to devise the means through which these can be controlled and to determine and predict the effect that control measures could have in reducing pollution sources.

Most of the environmental problems that occur due to pollutants are due to the movement of a pesticide away from the intended target location. Pollutants are transported in water from the soil surface to ground- and surface water. It is the capability of water as a solvent, which makes the carrying of significant quantities of dissolved material possible. In agricultural situations the most important solutes are nutrients (principally nitrogen compounds) and pesticides, which can enter water sources through surface run-off and/or leaching from both agricultural and non-agricultural land. Pollution of surface waters is chiefly the result of erosive surface run-off associated with intense rainfall or over-irrigation. Obviously, application of pesticides on or near surface water poses a more immediate threat. Groundwater pollution by pesticides is strongly linked to the rates at which the compounds are leached through the soil. The leaching rate is governed by the adsorptive capacity of the soil, the amount of overhead water, and the solubility of the compound leached.

Levels of pesticides detected in groundwater have generally been lower than those in surface water. However, the primary effort on regulating pesticide residues in water has been focussed on groundwater. The reason for this is that surface water undergoes severe chemical processing before its used as drinking water. So the primary aim behind the regulations is the protection of the larger proportion of the population (Gustafson, 1999). Groundwater on the other hand collected from wells internationally is used as is with little or no pre-treatment.

From an agricultural viewpoint, pesticide dissipation refers to losses that occur in the target area which relate to decreased biological activity on pests. However, pesticides that move away from the target area present particular problems in terms of potential environmental impacts. The discovery of pesticides residues in groundwater in the agricultural areas of the USA since the mid-seventies (Barbash and Reseck (1996), Walker and Hollis (1994)), has led to investigations into the processes that govern the movement of pesticides from their site of application.

Transport pathways can be broadly distinguished as Airborne or Waterborne. Locally, most of the investigations into pollution of the watercourse have been on the waterborne pathway. Waterborne pathways include runoff from farms, sub-surface percolation (leaching) and deliberate discharge of tail waters from holding ponds into rivers. However, waterborne pathways alone cannot account for all water contamination patterns.

The main mechanisms associated with the fate of pesticide applications are illustrated in Figure 3.1.

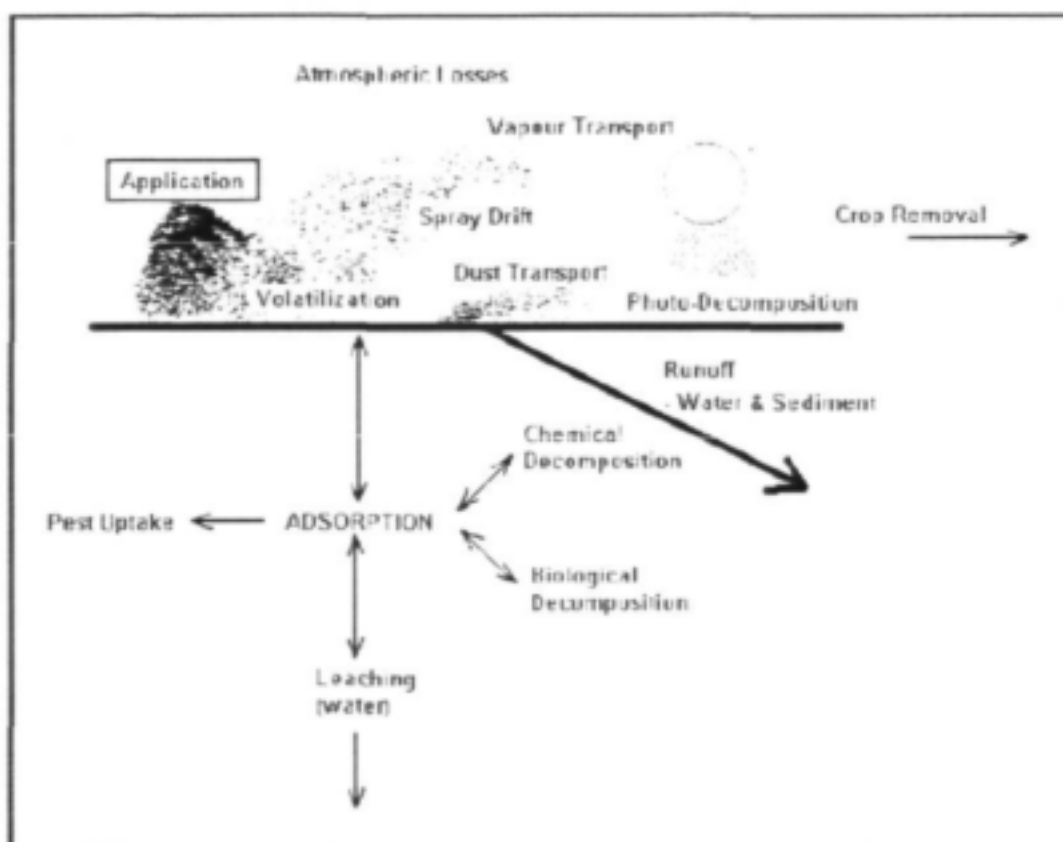


Figure 3-1 The main mechanisms associated with the fate of pesticide applications

What follows in Section 5, 6 and 7 is a discussion on each of the aspects as set out in Figure 3.1.

4. OVERVIEW OF THE EXTENT OF THE PROBLEM PESTICIDES

A comprehensive literature search was done for pesticides including data from all South Africa. This study overlapped with the Water Research Commission project No. K5/1326//1 on the identification and prioritization of groundwater contaminants in different hydrogeological environments in South Africa.

The bulk of the information was obtained from the proceedings of the European-Southern African conference organized in Cape Town (January 2003) on pesticides in non-target agricultural environments. A number of studies on measurements of pesticides in South African surface and groundwater were presented at this conference. Grange *et al* (2003) identified a large number of pesticide species and other organic compounds in surface water samples collected near Johannesburg and analysed with the Ion Composition Elucidation (ICE) technique. Naidoo and Buckley (2003) also carried out a comprehensive Water Research Commission project on pesticide wastes in South Africa and treatment options.

A summary of the literature review on pesticides in South Africa can be found in Table 4.1. In general, the most commonly detected and/or studied pesticides in South Africa were chlorpyrifos and endosulfan (mentioned 11 times in the literature), azinphos-methyl (10 times), atrazine, deltamethrin and penconazole (4 times). The number of times a pesticide was reported on in the literature is a very rough indication of its usage. However, it should also be noted that, in all literature studies, strong motivation was given as to why certain pesticide species were studied rather than others (e.g. it is known that certain pesticides are used on certain commonly cultivated crops in an area). For example, London *et al* (2000) selected the pesticides to be intensively monitored in the Western Cape (chlorpyrifos and endosulfan), according to the following criteria:

- Focus on critical pesticide agents, not on the full compound and by-products
- Industry opinion
- Spray programme recommended by the cooperative
- Health impacts based on international standards
- Usage in the specific study area as determined from expert industry opinion and from published literature
- Local spray applications of farmers, commercial spray operators and cooperatives regarding chemicals used (types, concentrations, methods of application) and weather conditions at time of sampling
- Likelihood of contamination based on industry expert opinion, U.S. Environmental Protection Agency (EPA) list of water pollutants and known groundwater ubiquity scores. The groundwater ubiquity score is based, amongst other factors, on the leachability of the pesticide that depends on its organic sorption coefficient and half-life in soil.
- Availability of analytical methodology for residues.

It is to be noted that, not only pesticides currently in use could cause an impact on the environment, but also pesticides that have been banned, but are characterised by prolonged persistence such as DDT. Furthermore, the review is concentrated on pesticide residue work that was conducted in the 1990 to 2003 time period. The residue data summarised from this research is based on ad hoc experiments, rather than monitoring experiments run over a extended time period. It must be kept in mind that there is severely limited pesticide monitoring data available for South African water systems and it is thus difficult to access data which would provide an indication of fluctuation in residue concentrations over time.

A general lack of data on groundwater pollution from organic contaminants was evident. This is due to:

- 1) Surface waters that are the main source of water supply in the country (little data is available on groundwater pollution).
- 2) Cost and difficulty in measuring organic contaminants.
- 3) Private companies that are often sensitive to making public data related to pollution problems available.

Table 4.1 Pesticides detected in South Africa

Type of source	Pesticides	Examples	References
Gardens and agricultural areas	1,2-dichlorobenzene	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	1,3-dichlorobenzene	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	1,4-dichlorobenzene	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	2-methylnaphtalene	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	2,4 D (2,4 DDD)	1) 5.5 ng L ⁻¹ in marine water and 0.6 ng g ⁻¹ in freshwater sediments in Eastern Cape. 2) Twenty-first most used pesticide in southern Africa in 1989.	Fatoki and Awofulu (2003) London and Myers (1995a)
Gardens and agricultural areas	2,6-dimethylphenol	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	4-chloro-3,5-dimethylphenol	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Aldicarb	1) Twenty-second most used pesticide in southern Africa in 1989. 2) High usage in Hex River Valley (Western Cape).	London and Myers (1995a) Weaver (1993)
Gardens and agricultural areas	Aldrin (banned)	0.6 ng g ⁻¹ in freshwater sediments in Eastern Cape.	Fatoki and Awofulu (2003)
Gardens and agricultural areas	Alphamethrin	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Agricultural areas and vineyards	Amitrole	Used until 20 years ago in southern Africa	London and Myers (1995a)
Gardens and agricultural areas	Anthracene	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Atrazine	1) Identified in surface water samples (Johannesburg). 2) Measured residues in water and fish. 3) Between 0.1 and 11.6 ug L ⁻¹ of atrazine and related triazines, depending on the season, in northern and central part of South Africa. 4) Regularly detected in surface and groundwater of the Vaalharts Irrigation Scheme.	Grange <i>et al.</i> (2003) Bouwman <i>et al.</i> (2003) Du Preez <i>et al.</i> (2003) Weaver (1993)
Agricultural areas and orchards	Azinphos-methyl	1) From 0.08 ug L ⁻¹ to 0.7 ug L ⁻¹ in water of constructed wetlands at Lourens River (Cape Town).	Bennett <i>et al.</i> (2003)

Type of source	Pesticides	Examples	References
		2) Loads of 27.8 g via runoff and 0.69 g via spray drift from 400 ha per event. 3) Measured predominantly in water samples. 4) 0.2 ug L ⁻¹ in Western Cape streams. 5) Detected in surface and groundwater of Grabouw (Western Cape). 6) 0.6 ug L ⁻¹ in water samples and 152 ug kg ⁻¹ in suspended sediments from Lourens river (Western Cape). 7) From 0 to 0.2 ug L ⁻¹ in water samples and from 0 to 27.9 ug kg ⁻¹ in suspended sediments from Lourens River (Western Cape). 8) Up to 0.38g L ⁻¹ in water samples and up to 244ug kg ⁻¹ in suspended sediments from Lourens River (Western Cape). 9) Thirteenth most used pesticide in southern Africa in 1989. 10) Detected in farm dams in Elgin, Western Cape. 11) High pollution likelihood.	Dabrowski and Schulz (2003a and b) Dabrowski <i>et al.</i> (2003a) Schulz and Dabrowski (2003) Dalvie <i>et al.</i> (2003), Solomons <i>et al.</i> (2003), London <i>et al.</i> (2000) Dabrowski <i>et al.</i> (2003) Dabrowski <i>et al.</i> (2002) Schulz <i>et al.</i> (2001) London and Myers (1995a) Davies (1997) London <i>et al.</i> (2000)
Agricultural areas and pome fruit trees	Azocyclotin	Eighth most used pesticide in the pome fruit industry in Western and Eastern Cape in 1989.	London and Myers (1995b)
Gardens and agricultural areas	Benomyl	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	BHC α - BHC	1) Measurement in fish tissue in Crocodile River, Mpumalanga. 2) 450 ng L ⁻¹ in marine water and 184 ng g ⁻¹ in freshwater sediments in Eastern Cape.	Heath <i>et al.</i> (2003) Fatoki and Awofulu (2003)
Gardens and agricultural areas	Biphenyl	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Bromopropylate	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Bromoxynil	Eighteenth most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	Camphor	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Captab	Tenth most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	Captan	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Carbaryl	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Carbofuran	Detected in surface and groundwater of the Vaalharts Irrigation Scheme.	Weaver (1993)
Agricultural areas and vineyards	Chlorfenapyr	Applied to grapevine (Western Cape).	Walton and Pringle (2003)
Agricultural areas and orchards	Chlorpyrifos μ	1) From 0.8 ug kg ⁻¹ to 12 ug kg ⁻¹ in sediment cores of wetlands	Bennett <i>et al.</i> (2003)

Type of source	Pesticides	Examples	References
and orchards		constructed along Lourens River (Western Cape). 2) Measured predominantly in suspended particle samples. 3) Measured in surface and groundwater of Hex River Valley, Grabouw and Piketberg (Western Cape). 4) Applied to grapevine (Western Cape). 5) 0.19 $\mu\text{g L}^{-1}$ in water samples and 245 $\mu\text{g kg}^{-1}$ in suspended sediments from Lourens River (Western Cape). 6) From 0 to 0.03 $\mu\text{g L}^{-1}$ in water samples and from 0 to 43.6 $\mu\text{g kg}^{-1}$ in suspended sediments from Lourens River (Western Cape). 7) <0.01 $\mu\text{g L}^{-1}$ in water samples and up to 344 g kg^{-1} in suspended sediments from Lourens River (Western Cape). 8) 19.13 $\mu\text{g L}^{-1}$ in water samples from Hex River valley (Western Cape). 9) Eleventh most used pesticide in southern Africa in 1989. 10) Detected in farm dams in Elgin, Western Cape. 11) High usage in Hex River Valley (Western Cape).	Dabrowski <i>et al.</i> (2003a) Dalvie <i>et al.</i> (2003), Solomons <i>et al.</i> (2003), London <i>et al.</i> (2000) Walton and Pringle (2003) Dabrowski <i>et al.</i> (2003) Dabrowski <i>et al.</i> (2002) Schulz <i>et al.</i> (2001) McGregor (1999) London and Myers (1995a) Davies (1997) Weaver (1993)
Gardens and agricultural areas	Citrus bands	Sixth most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Agricultural areas, orchards and vineyards	Copper oxychloride	1) Commonly applied as fungicide in South African vineyards and orchards. 2) Copper salts are the fourth most used pesticides in southern Africa in 1989. 3) High usage in Hex River Valley (Western Cape).	Snyman <i>et al.</i> (2003), London <i>et al.</i> (2000) London and Myers (1995a) Weaver (1993)
Gardens and agricultural areas	Cyfluthrin	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Cyhalothrin	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Agricultural areas, vineyards and sugarcane	Cypermethrin	1) Applied to grapevine (Western Cape). 2) High usage in Hex River Valley (Western Cape). 3) Effect on non-target organisms.	Walton and Pringle (2003) Weaver (1993) Leslie and Baxter (2003)
Gardens and agricultural areas	DDE	Measurement in fish tissue in Crocodile River, Mpumalanga.	Heath <i>et al.</i> (2003)
Malaria mosquito control	DDT (use restricted to malaria mosquito control)	1) Measured residues in water and fish. 2) Measured residues in surface waters in the Ubombo and Ingwavuma districts in	Bouwman <i>et al.</i> (2003) Sereda and Meinhardt (2003)

Type of source	Pesticides	Examples	References
	4,4-dichlorobenzophenone (oxidation product of DDT)	KwaZulu-Natal. 3) Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Malaria mosquito control, gardens and agricultural areas	Deltamethrin	1) Measured residues in surface waters in the Ubombo and Ingwavuma districts in KwaZulu-Natal. 2) Detected in surface and groundwater of Hex River Valley (Western Cape). 3) 1.4 ug L ⁻¹ in water samples from Lourens River (Western Cape). 4) High usage in Hex River Valley (Western Cape).	Sereda and Meinhardt (2003) Dalvie <i>et al.</i> (2003), Solomons <i>et al.</i> (2003), London <i>et al.</i> (2000) Dabrowski <i>et al.</i> (2003) Weaver (1993)
Gardens and agricultural areas	Diazinon	Tentatively identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Diazonin	Detected in KaalSpruit river (Midrand).	Papo and Mathebula (2003)
Gardens and agricultural areas	Dibenzofuran	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Dibutylphthalate	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Dichlorovos	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Tsetse fly control	Dieldrin (banned)	1) Measured residues in water and fish. 2) Measurement in fish tissue in Crocodile River, Mpumalanga.	Bouwman <i>et al.</i> (2003) Heath <i>et al.</i> (2003)
Gardens and agricultural areas	Diethylphthalate	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Dimethoate	1) Fifteenth most used pesticide in southern Africa in 1989. 2) High usage in Hex River Valley (Western Cape).	London and Myers (1995a) Weaver (1993)
Gardens and agricultural areas	Dimethylphthalate	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Dimethylpyridine	Tentatively identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Dinitro-ortho-cresol (DNOC)	Twentieth most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	Dinocap	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Diquat	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Dithiocarbamate	Extensively used fungicide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	EDB	Ninth most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas, orchards	Endosulfan a, b and sulphate	1) From 0.8 ug kg ⁻¹ to 12 ug kg ⁻¹ in sediment cores of wetlands constructed along Lourens River (Western Cape). 2) Measured predominantly in suspended particle samples. 3) 0.83 ug L ⁻¹ and 3.16 ug L ⁻¹ in surface and groundwater of Hex River Valley, Grabouw and Piketberg (Western	Bennett <i>et al.</i> (2003) Dabrowski <i>et al.</i> (2003a) Dalvie <i>et al.</i> (2003), Solomons <i>et al.</i> (2003), London <i>et al.</i> (2000)

Type of source	Pesticides	Examples	References
		Cape). 4) Applied to grapevine (Western Cape). 5) 0.35 ug L⁻¹ in water samples and 273 ug kg⁻¹ in suspended sediments from Lourens River (Western Cape). 6) From 0 to 0.3 ug L⁻¹ in water samples and from 0 to 34.75 ug kg⁻¹ in suspended sediments from Lourens River (Western Cape). 7) Up to 0.16 ug L⁻¹ in water samples and up to 245 ug kg⁻¹ in suspended sediments from Lourens River (Western Cape). 8) 1.794 ug L⁻¹ in water samples from Hex River valley (Western Cape). 9) Nineteenth most used pesticide in southern Africa in 1989. 10) Detected in 28 out of 29 farm dams in Elgin, Western Cape. 11) Low usage in Hex River Valley (Western Cape).	Walton and Pringle (2003) Dabrowski <i>et al.</i> (2003b) Dabrowski <i>et al.</i> (2002) Schulz <i>et al.</i> (2001) McGregor (1999) London and Myers (1995a) Davies (1997) Weaver (1993)
Gardens and agricultural areas	EPTC	Detected in surface and groundwater of the Vaalharts Irrigation Scheme.	Weaver (1993)
Gardens and agricultural areas	Ethylene dibromide	Extensively used nematocide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	Fenamiphos	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Fenarimol	Detected in surface and groundwater of Hex River Valley (Western Cape).	Dalvie <i>et al.</i> (2003), Solomons <i>et al.</i> (2003), London <i>et al.</i> (2000)
Gardens and agricultural areas	Fenthion	1) Detected in thorn bush ecosystems. 2) High usage in Hex River Valley (Western Cape).	Van der Walt and Badenhorst (2003) Weaver (1993)
Gardens and agricultural areas	Fenvalerate	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Folpet	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Formothion	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Agricultural areas and vineyards	Fosetyl-Al	Applied to grapevine (Western Cape).	Walton and Pringle (2003)
Gardens and agricultural areas	Glyphosate	1) Fifth most used pesticide in southern Africa in 1989. 2) High usage in Hex River Valley (Western Cape).	London and Myers (1995a) Weaver (1993)
Gardens and agricultural areas	Heptachlor (banned)	Measurement in fish tissue in Crocodile River, Mpumalanga.	Heath <i>et al.</i> (2003)
Gardens and agricultural areas	Hexaconazole	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Isobutylphthalate	Tentatively identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)

Type of source	Pesticides	Examples	References
Gardens and agricultural areas	Iprodione	1) Detected in surface and groundwater of Grabouw (Western Cape). 2) Extensively used in Western Cape. 3) High usage in Hex River Valley (Western Cape).	Dalvie <i>et al.</i> (2003), Solomons <i>et al.</i> (2003), London <i>et al.</i> (2000) London <i>et al.</i> (2000) Weaver (1993)
Gardens and agricultural areas	Lindane	Measurement in fish tissue in Crocodile River, Mpumalanga.	Heath <i>et al.</i> (2003)
Agricultural areas and vineyards	Mancozeb	1) Applied to grapevine (Western Cape). 2) Second most used pesticide in southern Africa in 1989. 3) High usage in the Hex River Valley (Western Cape).	Walton and Pringle (2003) London and Myers (1995a) Weaver (1993)
Gardens and agricultural areas	MCPA	1) Eighth most used pesticide in southern Africa in 1989. 2) High usage in Hex River Valley (Western Cape).	London and Myers (1995a) Weaver (1993)
Agricultural areas and vineyards	Mercaptothion	Applied to grapevine (Western Cape).	Walton and Pringle (2003)
Gardens and agricultural areas	Metaldehyde	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Agricultural areas and pome fruit trees	Methidathion	1) Sixth most used pesticide in the pome fruit industry in Western and Eastern Cape in 1989. 2) High usage in Hex River Valley (Western Cape).	London and Myers (1995b) Weaver (1993)
Gardens and agricultural areas	Methiocarb	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Metiram	Sixteenth most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	Mevinphos	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas, carrier of other pesticides	Mineral oil	Third most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	Naphthalene	Identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Nuarimol	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	o-hydroxybiphenyl	Tentatively identified in surface water samples (Johannesburg).	Grange <i>et al.</i> (2003)
Gardens and agricultural areas	Omethoate	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Oryzalen	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Paraquat	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Parathion	1) Used until 20 years ago in southern Africa. 2) Detected in farm dams in Elgin, Western Cape.	London and Myers (1995a) Davies (1997)
Gardens and agricultural areas	Paraquat	Fourteenth most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	Penconazole	1) Detected in surface and groundwater of Grabouw	Dalvie <i>et al.</i> (2003), Solomons <i>et al.</i>

Type of source	Pesticides	Examples	References
vineyards		(Western Cape). 2) Applied to grapevine (Western Cape). 3) Detected in farm dams in Elgin, Western Cape. 4) High usage in Hex River Valley (Western Cape).	(2003), London <i>et al.</i> (2000) Walton and Pringle (2003) Davies (1997) Weaver (1993)
Gardens and agricultural areas	Pirifeno	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Prochloraz	Seventeenth most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	Procymidone	1) 9 ug L ⁻¹ in water samples from Lourens River (Western Cape). 2) Low usage in Hex River Valley (Western Cape).	Dabrowski <i>et al.</i> (2003) Weaver (1993)
Gardens and agricultural areas	Profenofos	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Propetamphos	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Propineb	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas	Propoxur	High usage in Hex River Valley (Western Cape).	Weaver (1993)
Gardens and agricultural areas, orchards	Prothiofos	1) From 0.8 ug kg ⁻¹ to 12 ug kg ⁻¹ in sediment cores of wetlands constructed along Lourens River (Western Cape). 2) Detected in surface and groundwater of Hex River Valley (Western Cape). 3) High usage in Hex River Valley (Western Cape).	Bennett <i>et al.</i> (2003) Dalvie <i>et al.</i> (2003), Solomons <i>et al.</i> (2003), London <i>et al.</i> (2000) Weaver (1993)
Gardens and agricultural areas	Simazine	1) Seventh most used pesticide in southern Africa in 1989. 2) High usage and pollution likelihood in the Western Cape. 3) High usage in the Hex River Valley (Western Cape).	London and Myers (1995a) London <i>et al.</i> (2000) Weaver (1993)
Gardens and agricultural areas	Sulphur	Most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	Tartar emetic	Twelfth most used pesticide in southern Africa in 1989.	London and Myers (1995a)
Gardens and agricultural areas	Triadimefon	Low usage in Hex River Valley (Western Cape).	Weaver (1993)
Agricultural areas and pome fruit trees	Vanmidothion	1) Fourth most used pesticide in the pome fruit industry in Western and Eastern Cape in 1989. 2) High usage and pollution likelihood in the Western Cape.	London and Myers (1995b) London <i>et al.</i> (2000)
Gardens and agricultural areas	Vinclozolin	High usage in Hex River Valley (Western Cape).	Weaver (1993)

5. PESTICIDES IN THE ATMOSPHERE

The focus of this section is that of pesticides in the atmosphere once the spray has left the nozzle (atomisation). Aspects that are discussed include:

- Spray drift onto off-target receptor surfaces.
- Vapour transport of volatilised compounds (pesticides, herbicides, fungicides, etc.) into the atmosphere followed by dispersal and deposition of the vapour. The source may be during spraying (i.e. in-flight) and/or delayed (i.e. after deposition)
- Dust transport of contaminated dust, i.e. re-suspension.
- Factors and processes affecting atmospheric behaviour of pesticides

5.1 FACTORS AND PROCESSES AFFECTING ATMOSPHERIC BEHAVIOUR OF PESTICIDES

Losses of pesticides by drift, volatilisation and subsequent atmospheric transport have been increasingly recognised as a process, sometimes of major importance, in the loss of pesticides from the areas where they were originally applied. This process has, apart from limiting their effectiveness, the potential as a pathway for general environmental contamination. Pesticides could therefore potentially contribute to pollution of air, rain, soil, surface, and seawater.

This discussion will be limited to the mechanisms related to atmospheric losses. These include spray drift, volatilisation and dust transport. Perhaps the most widely studied and reported mechanism has been spray drift. The various factors that determine the extent of spray drift include particle sizes, the volatility of the pesticide (or herbicide), the method of application, nozzle spray angle, the distance between nozzle and target (boom height), spray pressure, relative humidity, ambient air temperature, wind direction, wind velocity and atmospheric stability.

A pesticide's vapour pressure is a measure of its tendency to evaporate or become a gas (volatilisation). Application losses of sprayed pesticides to the atmosphere can be significant; losses of 50 percent or higher have been measured for pesticides applied on windy days. Evaporation can continue after application (also known as vapour transport). What happens to pesticides lost to the atmosphere is not completely understood, but one fate is its return to the earth with precipitation.

The dust transport pathway operates by the windblown movement of pesticide bearing dust particles. The mechanisms for windborne dust include uplift during wind erosion episodes, vehicle entrainment on unpaved roads and during agricultural operations.

5.2 SPRAY DRIFT

Spray drift is the movement of a pesticide through air, during or after application, to a site other than the intended site of application. The affected area could be another crop, a garden or any other vegetation within range of the pesticide movement. Although in most cases, this movement is confined to the greater application domain, under certain conditions, off-target movement may affect areas at greater distances from the field.



Figure 5-1 Example of spray drift from a field sprayer

Drift is influenced by many factors that are usually grouped into one of the following four categories:

- Spray characteristics;
- Equipment and application techniques used;
- Weather conditions; and
- Operator care and skill.

The factors mentioned above are discussed below.

5.3 SPRAY CHARACTERISTICS

5.3.1 Spray Particle Size

Spray droplet size is by far the most important factor affecting drift. Since wind moves small droplets farther than large droplets, increasing droplet size can reduce spray drift. In general the longer the droplets remain airborne, the greater the chances of them being carried away by wind away from the application site. Small spray droplets are more susceptible to drift than the larger droplets because they tend to remain airborne much longer than the larger droplets.

The extent of drift under windy conditions also depends on the height of release above ground level. This is due to the vertical log-normal behaviour of the wind speed. Table 5.1 illustrates the effect of droplet size on the fall rate and carry distance in a well-defined wind speed.

Table 5.1 Influence of droplet size on potential distance of drift (Ross and Lembi, 1985)

Droplet Diameter (microns)	Type of Droplet	Time Required to Fall 3.3 m (10 feet)	Distance covered by droplets falling 3.3 m in a 4.8 km/hr (3 mph)
5	Fog	66 minutes	4.8 km
20	Very Fine Spray	4.2 minutes	30.5 m
100	Fine Spray	10 seconds	13.4 m
240	Medium Spray	6 seconds	8.5 m
400	Coarse Spray	2 seconds	2.6 m
1000	Fine Rain	1 seconds	1.4 m

Different effects are also witnessed in the vortices generated underneath aeroplane applications. An example of this is given in Figure 5.2, clearly illustrating the relatively unperturbed trajectory of a droplet of 800 μm in comparison to a fine spray of 100 μm diameter.

5.3.2 Vapour Drift

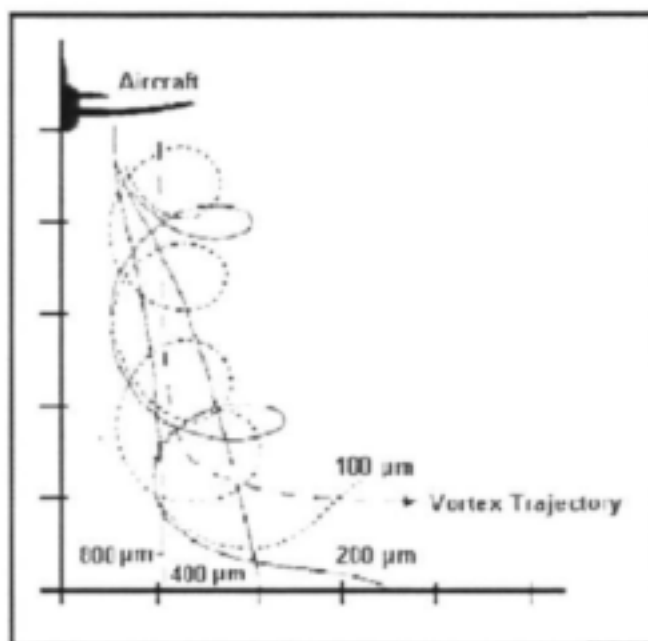


Figure 5-2 Example of aircraft vortex and droplet trajectories

Pesticides can drift as spray droplets. Some herbicides may be sufficiently volatile to cause plant injury from drift of vapour (fumes). Vapour drift occurs when a volatile herbicide (pesticide or fungicide) changes from solid or liquid into a gaseous state and moves from the target area. Vapour clouds may drift farther and over a longer period of time than spray droplets.

5.3.3 Equipment and Application Techniques

There are a number of techniques that can be applied to increase droplet size and reduce spray drift. These include reducing the spray pressure, increasing nozzle orifice size, using special drift reduction nozzles, include additives that increase spray viscosity, and rearward nozzle orientation in aircraft.

Many components of a sprayer can be adjusted to alter droplet size. Of these, nozzle type selection is one of the most critical. Spray droplets are produced from nozzles in different ways. A fan nozzle forces the liquid under pressure through an elliptical orifice and the liquid spreads out into a thin sheet that breaks up into different-sized droplets.

A flood nozzle deflects a liquid stream off a plate that causes droplets to form. A whirl chamber nozzle swirls the liquid out of an orifice with a circular motion and aids the droplet formation with a spinning force.

Various nozzle types and different spray pressures influence droplet sizes (Table 5.2). The full cone produces the largest droplets, which result in lower drift potential.

Table 5.2: Effect of nozzle type on droplet size at 2.7 atmospheres and 1.9 litres per minute (Spraying Systems Co. 1990)

Nozzle Type	Volume Median Diameter (VMD) (micron)
Hollow Cone	360
Extended Range Flat Fan	460
Standard Flat Fan	470
Full Cone	680

Note: Volume median diameter (VMD) is a term used to describe the droplet size produced from a nozzle tip. VMD is the droplet size at which one-half of the spray volume consists of large droplets and one-half consists of smaller droplets. Since it takes many more small droplets to make up one-half of the spray volume, there will always be more small droplets present in a typical spray pattern.

Spray pressure influences the size of droplets formed from the spray solution. The spray solution emerges from the nozzle in a sheet, and droplets form at the edge of that sheet. Increased nozzle pressure causes the sheet to be thinner, and this thinner sheet will break into smaller droplets. Also, larger orifice nozzles with high delivery rates produce a thicker sheet of spray solution and larger droplets than smaller nozzles.

The relationship between flow rate and pressure is not linear. For example, to double the flow rate would require the pressure to be increased four times. This action would greatly contribute to the drift potential and is not an acceptable method of increasing carrier volumes. Table 5.3 shows the mean droplet size for nozzles when spraying at three pressures. Higher pressures decrease the droplet size.

Table 5.3 Effect of spray angle and pressure on droplet size (Spraying Systems Co., 1990)

Pressure	Volume Median Diameter (microns)		
	1 atm	2.7 atm	4.1 atm
40°	900	810	780
65°	600	550	530
80°	540	470	450
110°	410	380	360

The nozzle spray angle is the angle formed between the edges of the spray pattern from a single nozzle. Nozzles with wider spray angles (110°) will produce a thinner sheet of spray solution, and smaller spray droplets than a nozzle with the same delivery rate but narrower spray angle (80°).

However, wide-angled nozzles are placed closer to the target than narrow-angled nozzles for proper overlap and the benefits of lower nozzle placement offsets the disadvantage of slightly smaller droplets for drift reduction.

The size or capacity of the nozzle also influences droplet size. The larger orifice increases the droplet size at a common pressure. Table 5.4 shows the influence of an increasing flow rate on droplet size at a constant pressure.

Table 5.4 Effect of flow rate on droplet size at 2.7 atm (Spraying Systems Co., 1990)

Volume Median Diameter (microns)		
Std. Flat Fan		
Extended Range 390	470	560
Flat Fan 360	460	560
Flood Jet 370	450	540
Full Cone	680	770

Orientation of nozzles is not critical for ground applications, but plays an important role in reducing drift from aircraft applications. When a nozzle is pointed backward toward the tail of the aircraft, larger droplets are produced. The same nozzle produces medium droplets when pointed downward and smaller droplets when pointed into the air stream.

A courser spray can be achieved by increasing the viscosity of the spray mix. Increased viscosity results in an upward shift of the droplet spectrum so that there are fewer of the fine drops that are subject to drift. Many spray additives can be added to a tank mix to increase its viscosity.

5.3.4 Atmospheric Conditions

Low relative humidity and/or high temperature will cause more rapid evaporation of spray droplets between the spray nozzle and the target. Evaporation also reduces droplet size, which in turn increases the potential drift of spray droplets. For typical applications with ground applicators, droplets of 50 microns and less will completely evaporate to a residual core of pesticide before reaching the target. Droplets greater than 200 microns will have no significant reduction in size before deposition on the target. Evaporation of droplets between 50 and 200 microns is significantly affected by temperature, humidity and other weather conditions. Some pesticide formulations are more volatile than others. However, low humidity may reduce the effectiveness of pesticides because rapid drying

of a spray droplet will reduce herbicide penetration into a plant. Also, plants growing in low humidity produce a thicker cuticle than in high humidity, resulting in greater resistance to herbicide penetration. In general, total drift movement of compound out of the target area will be greater with low relative humidity and high temperatures.

The amount of compound lost from the target area and the distance it moves will increase as wind velocity increases, so greater wind velocity generally will cause more drift. However, severe crop injury from drift can occur with low wind velocities, especially under conditions that result in vertically stable air. Wind is generally recognized as an important factor affecting drift, but vertical air movement often is overlooked. Normally, air near the soil surface is warmer than higher air. Warm air will rise while cooler air will sink which provides vertical mixing of air. Small spray droplets suspended in the warm air near the soil surface will be carried aloft and away from susceptible plants by the vertical air movement. A temperature inversion occurs when air near the soil surface is cooler than the higher air. Small spray droplets can be suspended longer, move laterally in a light wind and impact on plants two miles or more downwind. Temperature inversions are most common near sunrise and are generally associated with low wind and clear skies.

5.4 MODELLING DEVELOPMENTS AND CURRENT STATUS

During a spraying operation, some spray drifts off-target. The deposition of this drift onto a downwind receptor point depends on

- The extent of the sprayed area;
- The release height of the point source;
- The droplet size distribution;
- The meteorological conditions (wind speed, wind direction, and atmospheric stability); and
- The depositional characteristics of the receptor surface.

The droplet size distribution can be calculated from

- Method of application (e.g. aircraft);
- Nozzle characteristics, including
 - Orifice size;
 - Spray angle; and,
 - Nozzle orientation.
- Spray pressure;
- Viscosity altering additives;
- Application flow rate, and
- Volatility (evaporation tendency).

A number of approaches exist for the selection of a atmospheric dispersion model. Typical approaches include analytical approximations (Gaussian Plume and Gaussian Puff) or numerical solutions of the advection-diffusion equation and Lagrangian particle trajectory models. Gaussian models are so named because they assume the downwind distribution of a dispersing gas from a point source is normal or Gaussian when considering the concentration profile in the crosswind and vertical direction. These are the easiest to apply.

One of the earliest models used to describe spray drift was known as the FS Cramer-Barry-Grim (FSCBG) spray dispersion model (Teske and Curbishley 1990). This model is based on Gaussian modelling techniques developed by the United States Army. FSCBG could not describe the intricacies of the near-wake flow field that is critical to accurately describing pesticide deposition from aerial

spraying. Therefore, AgDISP was developed. The original ADISP stood alone and did not have many of the features of FSCBG such as extensive aircraft libraries, nozzle libraries (which associated spray nozzles with droplet size distributions), a canopy penetration algorithm and various other features. It was therefore decided to merge FSCBG and AGDISP. The resulting model retained the FSCBG name but with the inclusion of the near-field algorithm (Teske *et al* 1993).

The combined model interfaced the near-wake algorithm with the Gaussian algorithm to model dispersion from the near field to the far field. The interface required an arbitrary matching of the two approaches at some downwind distance. This resulted in what the development team referred to as the Aseam @ that manifested itself as a discontinuity in the deposition field. The model allows the user to turn off either algorithm and in time the AgDISP near-wake algorithm became the approach of choice for the entire deposition field. So, although FSCBG remained as the only readily available applied model to simulate pesticide deposition from aerial spraying, the core calculation typically being implemented had changed from Gaussian to the AgDISP near-wake computation in the majority of applications.

At the beginning of the 1990s, a pesticide drift modelling effort developed between a consortium of pesticide manufacturers and the United States Environmental Protection Agency (US EPA). The EPA required new pesticide formulations to be registered. As part of this registration process, field trials are run to measure the drift propensity of a given formulation. The leading pesticide manufacturers formed an association called the Spray Drift Task Force (SDTF) and began investigating modelling approaches. The SDTF adopted the AgDISP near-wake algorithm as the core algorithm for a drift simulation model for aerial pesticide spraying (Teske *et al* 1997). The most important contribution of the SDTF was a database derived from over 180 field deposition studies.

The SDTF also undertook an overhaul of the model interface and gave the model a modern 'look and feel' which is critical to user acceptance. The model that this effort produced is known as AgDRIFT.

Unfortunately, a detailed theoretical description of the AgDRIFT model could not be sourced via the Internet. It may be possible to find this through more traditional methods (i.e. hardcopy searches through international libraries).

As an alternative, Raupach *et al* (2001b) provided a detailed account of a relatively simple model based on mass conservation and a Gaussian model assumption. The article also provides an adequate account of removal mechanisms, including gravitational settling (terminal velocity) and removal due to impaction and Brownian diffusion. Workable relationships have been derived and included in the dispersion model. The model assumes a predefined droplet size distribution and does not predict this from given nozzle data. Furthermore, this model does not account for the distortion of pathways due to the vortex formed underneath an aircraft.

A critical input parameter in the dispersion model would be the particle size distribution and densities. In the case of highly volatile compounds, the particle size distribution could also be a function of time after release. The rate of evaporation would obviously determine the droplet size.

5.4.1 Vapour Transport Pathway

It was indicated previously that volatilisation could potentially remove a substantial amount of the total application. For example, Kennedy *et al* (1998) reported that about 70% of the total endosulfan deposited on crops is removed through volatilisation over a period of a few days after spraying. This vapour is dispersed by wind and may be deposited on downwind surfaces, including rivers. The vapour transport pathway therefore involves three sequential processes:

- a. Post-spray volatilisation of the pesticide from the crop;
- b. Dispersion of vapour by wind and turbulence; and
- c. Deposition of the vapour to water surfaces.

The first two processes (volatilisation and dispersion) determine the concentration in the air above the water surface, which is the forcing variable driving the third (deposition) process. Vapour transport is therefore driven by the difference between the concentrations in air and is a bi-directional process.

The pesticide dissolves from air into water when the air concentration is high and re-volatilises back to air when the concentration in air is low.

5.4.2 Summary Points for Model Development

It is not apparent whether the AgDRIFT model includes vapour transport from evaporating pesticides and herbicides.

Post-spray volatilisation may be described by a simple, exponentially decaying release rate constrained to conserve mass. A number of models describing the volatilisation of contaminants from soil and water to outdoor air have been published. Some examples are included in Jury (1984), Hwang and Falco (1986) and Raupach *et al* (2001b). The dispersion model could be the same as the model employed for spray drift. The difference would merely be in the removal terms (e.g. terminal velocity = 0).

The volatilisation model would require wind speed, atmospheric stability, surface roughness, and a number of chemical properties (e.g. diffusivity and viscosity).

5.4.3 Re-Suspension of Contaminated Dust Particles

The dust transport pathway operates by the windblown movement of pesticide bearing dust. The dust flux of the pesticide to a receptor is determined by the dust deposition flux and the mass fraction of pesticide on the dust.

From literature, it would appear that dust transport is not considered to be a significant pathway. However, taking into account the relatively dry South African conditions, this assumption may not be valid. The mechanisms that need to be included in the generation of airborne particulates are dust uplift during wind erosion events, uplift by vehicular traffic on unpaved roads, and uplift by agricultural operations. Models for these mechanisms have not been developed in South Africa. However, emission factors have been derived elsewhere (e.g. Cowher *et al*, 1988) and have been used extensively, typically in mining applications.

5.4.4 Proposed Model Specifications

Although existing models to describe the behaviour of herbicide and pesticide spray exist (most notably the AgDRIFT model), it is not known how easily these could be incorporated into a South African derived model that would include all pathways. Furthermore, the existing models primarily address near-field impacts (i.e. immediate carry-over to neighbours), whereas the objective in a South African context is also to include long-term and long-range impacts.

Adequate information is freely available in the national and international literature to develop a local model. These models have been peer reviewed and should therefore provide reasonably good performance. It is nonetheless understood that a validation of the proposed model is envisaged locally.

Figure 5.3 is a fairly rudimentary illustration of the proposed model. There would be a number of calculation engines, namely

- Source Definition Engine
- Near-Field Dispersion Engine
- Removal Mechanism Engine
- Re-Suspension Engine (including volatilisation of deposited material and rust entrainment)
- Medium to Long-Range Dispersion Engine

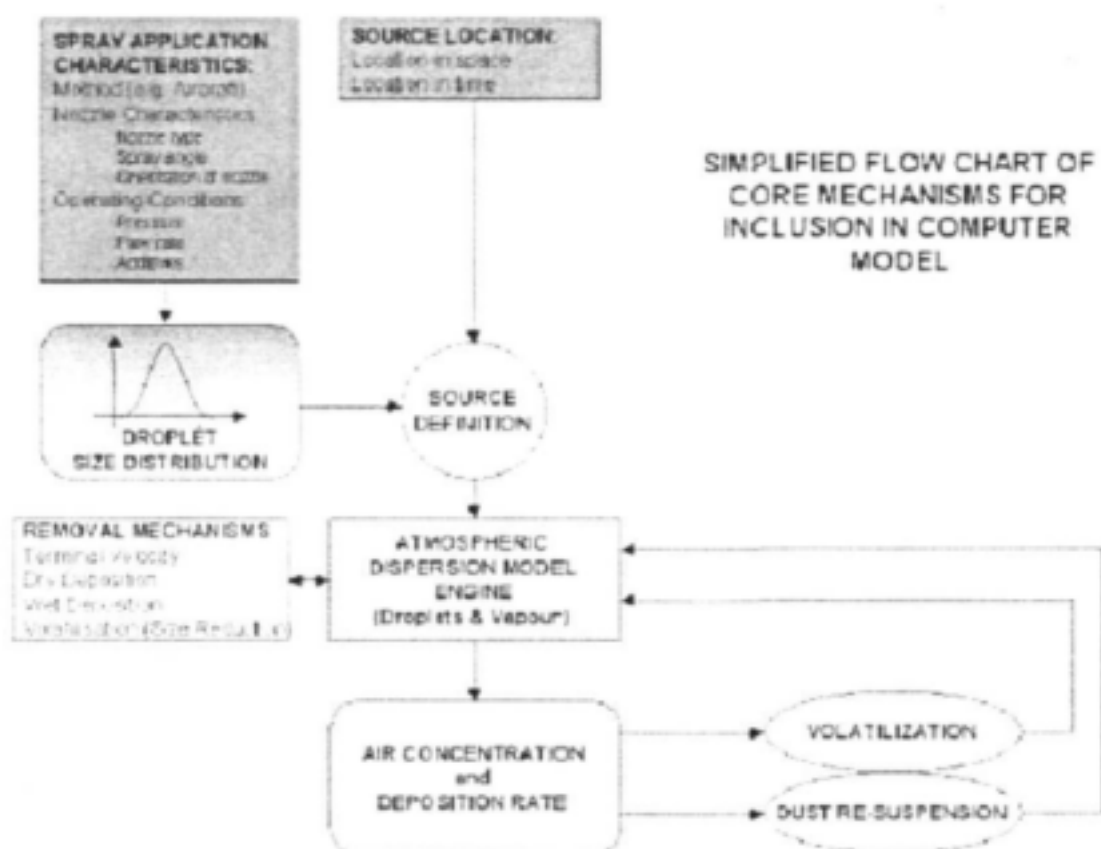


Figure 5-3 Conceptual design of pesticide spray drift and atmospheric dispersion model

The Source Definition Engine would be responsible for assigning droplet size distributions taking into account all nozzle information. This part of the program would also be responsible for locating the origin of the spray in terms of space and time coordinates (i.e. initial conditions). It is envisaged that the user would need to provide the physical properties of the pesticide, the nozzle description and the application mode (e.g. aircraft, ground based, etc.). Also to be provided at this level would be the spray plan (e.g. flight path).

The droplet size distribution would then be used as input into the Near-Field Dispersion Engine where the atmospheric dispersion simulations take place. Additional input parameters include meteorological

data (i.e. wind speed, wind direction, ambient air temperature, relative humidity, and solar radiation) and the wake effect caused by the aircraft (if applicable). Closely integrated with this engine is the Removal Mechanism Engine, which would be responsible for the removal through gravitational settling, dry deposition and volatilisation as the droplets travel through the air.

The release rate of pesticides into the atmosphere through subsequent evaporation of applied compound, or the entrainment of contaminated dust particulates is treated in the Re-Suspension Engine. These mechanisms are slightly divorced from the previous step, mainly because of the longer time steps involved. The rate of secondary vapour release is a function of wind speed, temperature and volatilisation potential. Secondary dust release is also a strong function of wind speed and the fraction absorbed onto the dust particle.

Medium to long-range dispersion of the total emissions (i.e. spray drift, vapour drift, secondary vapour and dust releases) would be treated using the Medium to Long-Range Dispersion Engine. This engine may be the same, or a subset, of the Near-Field Dispersion Engine.

The output from the model would typically include air concentration and deposition patterns for subsequent input into the other pathways.

5.5 CONCLUSIONS

Sufficient amount of information is freely available in the open literature describing the various processes involved in the atmospheric fate of pesticides. Although research level and commercial models are available, these may not easily be incorporated into the local Pesticide life-cycle model envisaged for South Africa.

A possible approach could be to use the locally developed HAWK dispersion model as a vehicle to attach certain of the modules (engines) described in Section 5.4.4. This would typically include the Source Definition Engine, components of the Near-Field Dispersion Engine (specifically addressing wake effects), Removal Mechanism Engine and the Removal Mechanism Engine.

The HAWK air pollution prediction system has been developed over the past fifteen years. The software code simulates the emission, transport and dilution of toxic chemical releases albeit routine or accidental. Airborne compounds may be gaseous or particulate. The system can operate in both (near) real-time and off-line (desktop studies). In the real-time mode, meteorological instrumentation transmits data continuously to a microprocessor where it is used to calculate the wind and turbulent fields. On-line emission rates, via in-stack analysers or mass balance calculations, can also be accommodated. Various mathematical models are incorporated to simulate the different types of release scenarios and varied pollutant types. Release scenarios could be continuous, intermittent or

instantaneous emissions. Furthermore, the source could assume various forms such as an evaporating pond, single or two-phase flow from storage tanks, chimneys, fugitive (volume or area) and line sources. The predicted concentrations are presented as isopleths (contours of equal concentration) on scanned background maps for easy orientation. These maps could be satellite images, contour maps, photographs or line drawings. In addition, the user can request vertical concentration profiles (i.e. concentration with altitude), horizontal slices through plumes and relative contributions from different sources at any location on the background maps. This is performed by simple "mouse" operations. The model incorporates a number of dispersion models, each applicable to the type of pollutant requiring simulations. The models range from analytical to numerical solutions.

5.6 RECOMMENDATIONS

It is both feasible and recommended that a model be developed, using internationally acceptable theory, which could be integrated into the Pesticide life-cycle model envisaged for South Africa.

It is further recommended to make use of the locally developed HAWK dispersion model as a vehicle to include the appropriate modules required to describe the mechanisms specifically related to the application of pesticides, herbicides and fungicides.

6. PESTICIDE RUNOFF FROM SOIL SURFACES

Non-point source agricultural pollution is generally considered one of the major threats to surface water quality in rural areas (Loague *et al.*, 1998). Nutrients, sediments and pesticides potentially enter aquatic environments and thus pose a risk to the communities that inhabit them (Cooper, 1993). Of all non-point source pollutants, pesticides are among the most crucial chemical stressors, simply because of their potential toxicity to many non-target aquatic organisms (Baier *et al.*, 1985).

In comparison to point sources, our knowledge of the extent of non-point source pollution is still quite restricted (Line, *et al.*, 1997). The main factors contributing to this situation are presumably the number and characteristics of variables (land-use, meteorology, soil types) that must be taken into account to understand and predict the occurrence and extent of non-point-source pollution (Walklate, 1992; Wauchope, 1996).

In South Africa only a limited number of studies have dealt with pesticide levels in farm dams (Davies and Peall, 1997; Hassett *et al.*, 1987), river ecosystems (Greichus *et al.*, 1977; Grobler, 1994; Roux *et al.*, 1994) or groundwater (John Weaver, CSIR, pers. communication). None of these studies attempted to establish a direct link between agricultural use of chemicals and contamination of the aquatic environment. Moreover, they have not addressed a specific non-point source as a potential route of entry into the freshwater ecosystem. A factor that has contributed to the lack of research is the shortage of laboratories with the equipment and expertise to carry out complex analyses (Dallas and Day, 1993).

More recently, extensive research has focused on non-point source pollution by pesticides in the Lourens River, Western Cape. After leaving a naturally vegetated fynbos area, the river runs through intensive farming areas (orchards and vineyards) in its middle reaches before flowing through the town of Somerset West. It has now been established that surface runoff is the most important non-point source of pesticide pollution in this catchment (Dabrowski and Schulz, 2003; Schulz, 2001a) and that most contamination of the Lourens River mainstream is as a result of runoff activity in the tributaries of the river (Dabrowski *et al.*, 2001; Schulz, 2001b).

Studies have shown that the first heavy rainfall after application results in the highest quantity of pesticides in surface waters (Domagalski *et al.*, 1997). Thus, in the context of the Western Cape, important periods for determining runoff-related contamination are at the beginning of April, when the first heavy rains normally fall after the end of the spraying season in late February (Dabrowski *et al.*, 2001; Schulz *et al.*, 2001; Schulz, 2001a). A further important sampling period is after heavy rainfall events during intense spraying periods occurring from October to December (Schulz, 2001a; Schulz, 2001b).

6.1 FACTORS AND PROCESSES INFLUENCING PESTICIDE RUNOFF

Surface runoff occurs when the precipitation rate exceeds the infiltration rate of the soil or when the length of a rainfall event exceeds the infiltration capacity of a soil. As a result, the quantity of pesticides that enter surface waters via runoff are dependent on a number of factors, the most important of which include; the time interval between the application of pesticides and the first heavy rainfall event; the slope and soil types of the catchment; the quantity of applied pesticide, as well as the size and characteristics of vegetated buffer strips (a zone of dense ground vegetation lying in between an agricultural field and a surface water body) (Cole *et al.*, 1997; Merkle and Bovey, 1974). Furthermore, runoff is highly dependent on the physicochemical properties of the pesticides as they determine the amount of pesticide available to surface runoff (Capel and Larson, 2001; Blanchard and Lerch, 2000). Pesticides can be found in two forms in surface runoff; in the soluble form (dissolved in

runoff water) and in the eroded form (sorbed to suspended solids). The proportion and quantity of a chemical in each form at a given site depends upon the extent of sorption with the associated soil matrix, partitioning between runoff water and the suspended eroded material in transit and the degradation rate in the soil. Thus, important physicochemical properties to consider include the water solubility, half life (DT_{50}) and soil sorption index i.e. the tendency of a chemical to bind to organic carbon in the soil (K_{OC}). Pesticides with low water solubilities tend to be more associated with suspended sediments as opposed to being dissolved in the water phase of the surface runoff. A study which monitored the contamination of the Lourens River due to surface runoff input of pesticides (Dabrowski *et al.*, 2001), showed that 2 out of 23 samples had detectable concentrations of chlorpyrifos (an organophosphate insecticide with low water solubility [1.2 mg/l] and a high K_{OC} [6070]). In the same study, almost all sediment samples (10 out of 15) were found to contain high concentrations of chlorpyrifos (up to 150 µg/kg). In contrast, azinphos-methyl (high water solubility [28 mg/l] and low K_{OC} [1000]) was found almost exclusively in water samples, with very few sediment samples containing detectable pesticide residues. Thus the water solubility and K_{OC} of a compound is very important in terms of the fate of the pesticide and its partitioning in the water body. As a result, when monitoring for runoff related pesticide contamination, it is important to sample for pesticides in both the water and suspended sediment phases. Soil properties also play an important role in terms of the amount of surface runoff and the quantity of transported pesticide that occurs during an event. Soils with higher organic carbon percentages will tend to bind pesticides more than soils with low organic carbon content, while for any given precipitation amount, loamy and clay soils will give rise to a greater quantity of surface runoff (and hence pesticide loss) than a sandy soil (Reus *et al.*, 1999). Slope has been shown to be the most important factor influencing runoff but can be greatly modified by the presence of vegetation (Wilcox and Wood, 1989). The timing of rainfall events in relation to application date and half life of pesticides also plays a significant role in determining the amount of pesticide available in surface runoff. Heavy rainfall events occurring a few days after the application of pesticides have resulted in the detection of high concentrations of pesticides in the Lourens River (Schulz, 2001b). Pesticides with long half-lives (i.e. organochlorines such as endosulfan and DDT) can be measured in surface runoff many months after the last application (Dabrowski *et al.*, 2001). Therefore a complex interaction of a multiple number of variables ultimately influences the quantity of pesticide that can be expected to occur in surface runoff.

6.2 IMPORTANCE OF RUNOFF

Because of the direct input and bio-availability of pollutants so introduced, spray drift is commonly regarded as the worst case scenario for pesticide exposure in aquatic risk assessment (Ganzelmeier *et al.*, 1995; AEDG, 1992; Gilbert and Bell, 1988). In the few studies that have however directly compared these routes of entry in the same catchments, runoff has been shown to be the most important non-point source of pesticide pollution (Dabrowski and Schulz, 2003; Schulz, 2001a; Kreuger, 1998) and is regarded as the most important factor in terms of contamination of surface waters in arid areas (Everts, 1997). Since many insecticides have a relatively low water solubility, their transport via leaching is negligible (Flury, 1996). Moreover, leaching would contribute preferentially to contamination of groundwater and thus might only affect surface waters under specific geological conditions (Squillace *et al.*, 1996; Zullei-Seibert and Skark, 1993). Consequently spray drift and edge-of-field runoff are commonly regarded as the most important routes of entry for agricultural non-point source insecticide pollution into surface waters (Groenendijk *et al.*, 1994) but may result in considerably different exposure scenarios (Erstfeld, 1999). Whereas spray drift leads to direct input of pesticides dissolved in the water phase, contamination during runoff is often largely as a result of pesticides associated with suspended particles (Mian and Mulla, 1992). Although pesticides dissolved in the water phase pose more of an immediate toxicological threat, sediment associated pesticides have also been shown to adversely affect macro-invertebrate communities (Schulz and Liess, 2001).

A study in the Lourens River showed that surface runoff leads to higher loads and concentrations of aqueous azinphos-methyl (high water solubility); both on an event and annual basis. One potential reason is related to land-use factors and the fact that the spatial and temporal scale of runoff contamination is far greater than spray drift. During a runoff event each tributary within a catchment can receive surface runoff simultaneously, resulting in a high potential input of pesticides. While vegetative buffer strips can reduce pesticide loss (Patty *et al.*, 1997), the presence of erosion rills may jeopardize their positive effect and can thus result in rapid pesticide input (Castelle *et al.*, 1994). Furthermore, rainfall is non-directional and does not restrict the number of orchards susceptible to runoff. Spray drift on the other hand is dependent on downwind orientation of water bodies to the point of application (Longley and Sotherton, 1997) and can be prevented by windbreaks (Adriaanse, *et al.*, 1997). Thus, land-use and meteorological conditions potentially restrict the number and length of tributaries that can possibly be affected by spray drift.

Furthermore, owing to the long half-life of some pesticides, runoff-induced pesticide loading can occur long after the most recent application (Kreuger, 1998; Williams *et al.*, 1995) and therefore integrates chemical input over a prolonged period of time. In comparison, spray drift is instantaneous and contamination can only occur during application, in combination with specific meteorological requirements, which further restricts the potential for contamination.

It must be noted however, that the findings of this study may be specific to the land-use of the Lourens River catchment and the fact that there are a high number of windbreaks in the catchment. As mentioned previously, the physicochemical properties of pesticides also play an important role, and other more insoluble pesticides with a higher adsorption co-efficient may lead to spray drift being a more important route of aqueous dissolved pesticide exposure, and runoff contributing mainly to particle associated contamination.

6.3 MODELLING DEVELOPMENT AND CURRENT STATUS

As evidence of the detrimental effects of pesticides in aquatic environments accumulates, so the need to accurately predict areas of risk becomes more and more urgent (Black *et al.*, 2000). This has necessitated the implementation of risk assessments so as to identify problem areas and any possible effects on workers, consumers, the environment and non-target plants and animals. Assessment of potential side effects of pesticide use includes an estimation of exposure, which is usually based on predicted environmental concentrations (PECs) (Bascietto *et al.*, 1990). However, it has also been pointed out that modelling approaches in exposure assessment should also be validated through measurement of real contamination in the field (Solomon *et al.*, 1996; Wauchope, 1996).

Standardised aquatic risk assessments typically involve an assessment of exposure (e.g. fate and behaviour of a substance in the environment) and ecotoxicological effect (based on single species toxicity tests covering the main biological components of aquatic systems: *Daphnia*, fish and algae). Environmental fate models have been used to describe the behaviour of crop protection products since the early 1980s and the models presently play an integral role in pesticide regulation and registration in Europe and the USA (Jones and Mangels, 2002). In the European Union exposure assessments have become more detailed and have incorporated the use of fate models to calculate PECs in an attempt to determine potential environmental loadings. The registration procedure for plant protection products according to the Council Directive 91/414/EEC includes the use of models for the calculation of PECs in surface waters (FOCUS, 1997). Depending on the PECs, further investigations (e.g. toxicity tests) have to be conducted in order to demonstrate acceptable risk to aquatic organisms. Thus models currently exist which make calculations for each of the main non-point sources of pesticide contamination of surface waters; spray drift (Spray Drift Calculator), drainage (MACRO) and surface runoff (PRZM) (FOCUS, 2001). Additional fate models are also used to predict the fate of chemicals in surface waters as a result of dissipation, partitioning and prevailing

hydrological conditions (e.g. TOXSWA in the EU and EXAMS in the USA). Pesticide fate models are also similarly used in the USA according to the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA). Geographical Information Systems (GIS) have played an integral role in the application of these models in analysing various spatial, topographical, pedological, hydrological and land-use input variables required for the models.

Because of the combination of land-use, meteorological and chemical factors, the prediction of runoff requires a great number of input variables and complex models such as PRZM, CREAMS, GLEAMS and AgNPS have been developed for this purpose (Donigan Jr. and Huber, 1991). These models have been validated in the field with some success (Jones and Magels, 2002; Grunwald and Norton, 2000; Solomon *et al.*, 1996; Donigan Jr. and Huber, 1991), but have disadvantages in that they are complicated and normally require a large amount of input variables that may not be generally available (Adriaanse *et al.*, 1997). Minimal data requirements and ease of application are the main advantages of simpler simulation methods (Donigan Jr. and Huber, 1991).

6.4 MODELLING RUNOFF RELATED PESTICIDE CONTAMINATION IN THE LOURENS RIVER CATCHMENT

Based on previous studies of runoff-induced pesticide pollution in the Lourens River catchment two modelling approaches were implemented using a GIS-based runoff formula by Reus *et al.* (1999). This formula is designed as a simple tool for prediction of pesticide loss in runoff and has been proposed as a risk indicator for runoff by the Organisation for Economic and Co-operative Development (OECD). The aim of the first approach (Dabrowski *et al.*, 2002) was to predict relative runoff induced insecticide loss from sub-catchments of the Lourens River using the runoff formula by Reus *et al.* (1999), and to validate the results by comparison to measured levels from field samples taken in the tributaries of the sub-catchments during runoff events. Through using the formula the intention was not to accurately predict exact pesticide concentrations in the tributaries. Rather, it was to determine whether the formula predicts differences in the average loss of pesticides during a runoff event, based on catchment variables (i.e. soil type, TOC% of the soil, slope, precipitation and buffer strip width) and the physicochemical characteristics (i.e. half-life and K_{OC}) of applied pesticides in the sub-catchments (which are incorporated into the formula), and whether these differences are reflected in water samples taken at all of the sites. The study focussed on three pesticides: azinphos-methyl (relatively water soluble and low K_{OC}), chlorpyrifos and endosulfan (very low water solubilities with high K_{OC} values). The model was also adapted so as to predict particle-associated pesticide contamination.

Correlations showed a positive relationship between the predicted average loss and observed average concentrations for all pesticides in both the water and sediment phase in each sub-catchment. This indicates that the loss of pesticides calculated by the formula is well linked to the level of contamination in the tributaries of the sub-catchments. The formula predicted a high average loss for azinphos-methyl and lower values of average loss for chlorpyrifos and endosulfan (the lowest predicted loss per sub-catchment) in the water phase. This is in agreement with the partitioning of azinphos-methyl according to the samples taken during runoff and to the low K_{OC} of this pesticide. Predictions for azinphos-methyl were better than for chlorpyrifos and endosulfan in the water phase, which is most probably attributed to the fact that chlorpyrifos and endosulfan were not detected as frequently or in as high concentrations (due to the low water solubility) resulting in poorer resolution of relative contamination amongst the sub-catchments. The predictions for the sediments were generally much lower than the measured levels in the field.

The original use of the formula is intended to predict the loss of insecticide during a specific event at a specific site, but in the context of this study it was used as a general predictive tool for comparisons between sub-catchments. A general limitation of the model in this respect is that it does not predict pesticide loads or concentrations in the tributaries. Accurate predictions of concentrations would

require measurement of turbidity and discharge with high temporal resolution over the course of a runoff event for each tributary as well as the duration of the runoff event. Alternatively, predicted loads would need to be validated by loads of pesticides measured in the tributaries, which would also require the same detailed field measurements. Whilst the model did not predict accurate concentrations of pesticides in the tributaries, it was able to predict differences in the degree of contamination of surface runoff between sub-catchments. This approach can therefore potentially aid farmers or environmentalists in management planning and in identifying and prioritising areas of risk.

The second approach (Dabrowski and Schulz, 2003) focused on predicting dissolved azinphos-methyl contamination by runoff, with the aim being to predict accurate loads as a result of runoff activity. Monitoring of azinphos-methyl took place together with detailed measurements of discharge and an intimate knowledge of spraying programmes. In this case, predicted loads in the tributaries and mainstream Lourens River accurately represented measured loads (between a factor of 0.81 and 1.34 difference), which indicated the suitability of the GIS-integrated formula for South African orchard areas.

The formula used in these studies has a theoretical basis, but also comprises empirical and physical components; a hydrological model (which takes rainfall and soil characteristics into account), catchment related factors which influence the extent of runoff (slope, plant interception and buffer strips), a first order kinetic for the fate of a pesticide (half-life) and a term referring to the proportion of pesticide occurring in the water phase of runoff (physicochemical properties of the pesticide). This formula has also been successfully used in other studies in Italy (Verro, *et al.*, 2002) and Germany (Ralf Schulz, Syngenta U.K., pers. comm.) and comprises the following variables:

$$L\%_{\text{runoff}} = (Q/P) \times f \times \exp(-3 \times \ln 2 / DT_{50\text{soil}}) \times 100 / (1 + K_d)$$

where: $L\%_{\text{runoff}}$ is the percentage of application dose being available in runoff water as a dissolved substance, Q is the runoff amount (mm) calculated according to the model of Lutz (1984) and Maniak (1992), P , the precipitation amount (mm), $DT_{50\text{soil}}$ the half life of active ingredient in soil, f the correction factor, with $f = f_1 \times f_2 \times f_3$, where f_1 the factor that reflects the influence of field slope on $L\%$ if slope is $< 20\%$, where $f_1 = 0.02153 \times \text{slope} + 0.001423 \times s24$, f_2 factor that reflects the influence of plant interception (PI) percentage on $L\%$, where $f_2 = 1 - \text{PI}/100$, f_3 the factor that reflects the influence of a densely covered buffer zone on $L\%$, where, $f_3 = 0.83^{\text{WBZ}}$ and WBZ is the width of buffer zone (meters); if the buffer zone is not densely covered with plants, in which case the width is set to zero, K_d the $K_{OC} \times \%OC/100$ where K_{OC} is the sorption coefficient of the active ingredient to organic carbon and $OC\%$ is the organic carbon content of the soil.

6.5 OTHER MODELS

6.5.1 Pesticide Root Zone Model (PRZM)

Of the various number of runoff models presently existing, PRZM is probably the most widely used (FOCUS, 1997) and has been validated successfully via field based measurements (Jones and Mangles, 2002; Singh and Jones 2002). It is used in the EU risk assessment process (FOCUS, 2002) and is a standard model to be used in environmental risk and exposure assessments by the United States Environmental Protection Agency (U.S. EPA). The model is included in the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) list of recommended regulatory models for USA Pesticide Registration.

PRZM is a management model, developed in US by the EPA, which allows the user to perform dynamic simulations of the fate of pesticides. It is a one-dimensional, field, daily time scale model that can be used to estimate runoff, leaching and associated pesticide loading. PRZM uses an SCS (Soil

Conservation Service) efficient method for determining the approximate amount of runoff from a rainfall event and the Universal Soil Loss Equation to estimate erosion (developed by the United States Department of Agriculture). The amount of infiltrating water is determined by subtracting runoff, evapotranspiration and interception losses from rainfall. Subsequent routing of soil water within the root zone is based upon a "tipping bucket" scheme that requires the specification of field capacity and wilting point. Evapotranspiration losses are divided between evaporation from crop interception, evaporation from the soil and crop transpiration. Potential evaporation is calculated by the input of pan evaporation and by using a pan factor. PRZM does not simulate sub-surface lateral flow, macropore flow, by-pass flow or tile drainage. PRZM is quite flexible in the way that pesticide application can be specified, to the soil or to plant foliage; the model also allows for the specification of some types of tillage operation and irrigation. Dissolved, adsorbed and vapour phase concentrations in the soil are estimated by simultaneously considering the processes of pesticide uptake by plants, surface runoff, erosion, decay, volatilisation, foliar wash-off, convection, dispersion and sorption. Pesticide degradation is described using first-order kinetics, without dependence on temperature or moisture content and pesticide sorption is modelled as linear. The model requires the input of a large number of parameters, some of which are difficult to measure, such as hydrodynamic dispersion coefficient, enthalpy of vapourisation and the depth to which evapotranspiration was extracted from the soil profile (Trevisan *et al.* 2000).

6.5.2 Groundwater Loading Effects of Agricultural Management Systems (GLEAMS)

GLEAMS is a continuous simulation, field scale model, which was developed as an extension of the Chemicals, Runoff and Erosion from Agricultural Management Systems (CREAMS) model. GLEAMS assumes that a field has homogeneous land use, soils, and precipitation. It consists of four major components: hydrology, erosion/sediment yield, pesticide transport, and nutrients. GLEAMS was developed to evaluate the impact of management practices on potential pesticide and nutrient leaching within, through, and below the root zone. It also estimates surface runoff and sediment losses from the field. GLEAMS was not developed as an absolute predictor of pollutant loading. It is a tool for comparative analysis of complex pesticide chemistry, soil properties, and climate. GLEAMS can be used to assess the effect of farm level management decisions (i.e. planting dates, cropping systems, irrigation scheduling, and tillage operations) on the potential for chemical movement.

The model also accounts for varying soils and weather in determining leaching potential. GLEAMS can also be useful in long-term simulations for pesticide screening of soil/management. The model tracks movement of pesticides with percolated water, runoff, and sediment. Upward movement of pesticides and plant uptake are simulated with evaporation and transpiration. Degradation to metabolites can be simulated for compounds that have potentially toxic by-products. Erosion in overland flow areas is estimated using a modified Universal Soil Loss Equation. In order to simulate the many events occurring on a field, the model is divided into three separate sub-models, or parameter files. These sub-models include hydrology, erosion/sediment yield, and chemical transport. The hydrology component simulates runoff due to daily rainfall using a modification of the SCS curve number method. Hydrologic computations are determined using a daily time step. Different topographic configurations and surface flow processes may be selected to properly assess the sediment detachment and deposition on the land surface. The chemistry component of the GLEAMS is divided into pesticide and nutrient sub-models. The pesticide component of the GLEAMS incorporates the surface pesticide response of CREAMS with a vertical flux component to route pesticides into, within, and through the root zone. Characteristics of pesticide adsorption to soil organic carbon are used to partition compounds between solution and soil fractions for simulating extraction into runoff, sediment, and percolation losses. Pesticide dissipation in soil and on foliage is treated as a first-order process with a different apparent half-life for each.

7. SOIL MOBILITY, BEHAVIOUR AND FATE

7.1 PROCESSES AND FACTORS INFLUENCING SOIL MOBILITY AND FATE

This section of the document has been collated from the following sources namely: Weaver and Reinhardt, 1999 (in Conrad *et al* 1999) and Meinhardt 2003 with permission of the authors.

There are many factors that determine the rate of dissipation or loss of a pesticide from soil. Basically, two distinct groups of factors are involved, namely transfer and transformation processes. Transfer processes involve the transferral of a compound from point A to point B, without its biological activity necessarily being affected. Transformation, on the other hand, entails the changing of the chemical structure of molecules, which usually results in loss of biological activity. Transformation of a compound may occur at any stage during its transfer from one location to another.

The fate of a pesticide applied to soil is governed to a large extent by its adsorption on soil colloids, desorption of the adsorbed fraction, its degradation and persistence. A portion of the amount of a pesticide that is sprayed onto plants can be expected to eventually reach the soil through wash-off by rain, or simply, through the spray solution missing the target (the crop) and landing directly on the soil surface. Pesticides dissipate from soil in many ways. Dissipation infers that the chemical is transformed into by-products, which are further transformed. In addition, the parent molecule and its by-products are also transferred to other places and transformed until the compound disappears or is no longer detectable. The many processes in the dissipation of pesticides are categorised as transfer processes or transformation processes. Some of the more important routes of dissipation are chemical degradation, leaching, microbial degradation, uptake by living organisms, photo-decomposition and volatilisation. There are numerous reviews on the chemical, physical and biological processes involved in the dissipation of pesticides in the environment (Bollag and Liu, 1990; Wolf, Mingelgrin and Miller, 1990; Weber, 1991a, b; Leeds-Harrison, 1995; Walker, Welch and Turner, 1995).

The physics of soil water and solute movement can be used to determine the fate of pesticides. The physico-chemical interactions of the solute with soil particle surfaces will vary depending on the nature of the solute as well as the solid surfaces it encounters. Partitioning of solutes between the solid and water fractions of soil will determine the extent and duration of their presence in one or the other soil compartment. The partitioning is governed by the relative affinity of the pesticide for soil components. In general, solutes occurring mainly in the water phase of the soil would be prone to leach, whereas those compounds that are associated with the solid phase (soil particles) will be less exposed to this form of dissipation. Thus the more soluble the compound the greater the potential for the compound to leach. The mechanisms of water and solute movement to surface and groundwater have been reviewed (Enfield and Yates, 1990; Leonard, 1990; Leeds-Harrison, 1995; Leake and Gatzweiler, 1995), and time-dependent sorption processes in soil by (Koskinen and Harper, 1990; Walker *et al*, 1995).

Logically, compounds that are chemically stable, and resistant to microbial degradation, will tend to persist for prolonged periods of time in soil. Hence the risk that a persistent compound may leach and eventually contaminate groundwater is high. Persistence and mobility in soil are therefore important predictors for estimating the potential risks for the principal users of such waters, i.e. plants, animals and humans. Risk may be expressed as a function of both the inherent toxicity of a compound and a sensitive organism's exposure to it:

$$\text{Risk (Hazard)} = \text{Toxicity} \times \text{Exposure}$$

What this means is that an organism may be harmed when exposed for only a brief period to a highly toxic compound, and that a low toxic product is unlikely to cause injury in the short term, but possibly in the long-term.

The risk posed by pesticides in terms of contamination of groundwater resources can be described as follows:

$$\text{Risk} = \text{Persistence} \times \text{Mobility}$$

Reviews on the impact of pesticides on the environment (Madhun and Freed, 1990), and on risk assessments/interpretation (Severn and Ballard, 1990; Zimdahl, 1993) exist.

7.1.1 Transfer Processes

Transfer processes involve the following mechanisms:

- adsorption on, and desorption from, soil colloids;
- uptake by plants (absorption);
- exudation and retention by living organisms;
- movement on the soil surface (run-off);
- volatilization;
- leaching (downward movement in soil), or
- capillary movement (upward transport in soil).

Comprehensive reviews on the effects of soil factors on herbicide activity have been presented by Riley and Morrod (1976), Eagle (1976), Walker (1980), Hance (1983) and Weber (1991a, b). Soil factors could be expected to have similar effects on pesticides with the same chemo-physical characteristics, because their mobility in and uptake from the soil depends on the soil/pesticide interaction. Differential biological activity (bioactivity) of pesticides is only expressed once the chemical reaches the active site in sensitive organisms.

7.1.2 Absorption, exudation and retention by plants

The amount of pesticide required to affect (harm) target species is determined by:

- The rate of uptake (absorption) by the organism.
- The release of absorbed compounds through exudation.
- The rate of metabolism (inactivation) of a compound inside the organism.
- The inherent sensitivity of the organism towards a pesticide.

Absorption of pesticides by plant roots reduces the amount that is available to leach and possibly reach free water beyond the rooting zone. The extent to which pesticides are removed from soil by plant roots is dependent on the physico-chemical characteristics of the compounds and the soil. Pesticides that are mobile in both the soil and the plant system will get to the target sites in the plant quickest. This trait is of particular importance for herbicides. A high degree of water solubility will assist in the rapid transport of compounds inside plants.

It is virtually impossible to predict how much of an applied pesticide is actually absorbed by target organisms. According to Weber (1991a, b) the amount of a mobile herbicide that is normally absorbed and retained by plants ranges from one-tenth to five per cent of the total amount applied. High plant numbers may result in much larger amounts being removed from the root zone of soil, but amounts

rarely exceed 10% of that applied. The fraction which is absorbed is sooner or later detoxified or compartmentalised in the plant. Thus, absorption of pesticides from the root zone is expected to contribute little to their dissipation from the soil sub-surface.

7.1.3 Adsorption on soil colloids

The rate of movement of a solute form is dependent on its sorption capacity to the soil and is thus an important process in determining the fate of pesticides in soil (Bailey and White, 1970). Adsorption of the solute in the soil is a function of chemical characteristics of the solute, but also soil characteristics. Soil chemical and physical properties, the chemical structure of the compound, and environmental factors such as temperature and soil water content are important. Solutes are adsorbed to soil colloids (Mora *et al.*, 1996) which carry an electrical charge and which adsorb solutes carrying a charge. The adsorption of solutes to colloids is a function of the charge that the molecule carries, and the charge on the colloid. Solutes are present in soil solution either as anions, or cations, or neutral molecules. Amphoteric compounds carry a positive and negative charge. Clay minerals carry a negative charge and attract mainly cations whereas organic material has an affinity for anions, cations and neutral molecules. This is why the organic matter content of the soil has such a large influential effect on the mobility of pesticides in soil.

Adsorption, and not water solubility of a compound *per se*, is the major factor determining pesticide mobility in soil. Adsorption is generally temporary as expected in an adsorption/desorption system. Equilibrium exists between the fraction of solute adsorbed and that in soil water. Adsorption also reduces the availability of compounds for absorption by plants for instance, thereby reducing their immediate bioactivity. The availability of pesticides in soil for absorption by plant roots or other organisms occurring in the soil is dependent on the adsorption-desorption equilibrium being shifted towards the desorption side.

This equilibrium is variable due to the involvement of the different, often interrelated, constituents of the solid soil phase. These soil constituents include clays, hydrous oxides of iron and aluminium, organic material and various other organic substances. It is likely that due to the dynamic and open nature of soil, absorption by plants, insects and micro-organisms, and degradation processes ensure that equilibrium with respect to pesticide concentration between the adsorbed and solution phases is never reached.

It has been shown that herbicides generally, and the triazines particularly, are reversibly adsorbed on the organic matter and clay mineral fractions in soils (Frissel, 1961; Talbert and Fletchall, 1965; Weber, 1970a,b; Hayes, 1970; Schiavon, 1988). In spite of relatively low organic matter contents in South African soils, Smit, Nel and Fölscher (1980), Nel and Reinhardt (1984) and Ehlers, Reinhardt and Nel (1987, 1988) found that the organic carbon content (% C) of soil was strongly and negatively correlated with atrazine bioactivity. In all the above-mentioned studies, organic matter content consistently predicted atrazine availability better than either cation exchange capacity (CEC), total clay content or individual clay minerals. According to Eagle (1983b), a positive relationship usually exists between the clay and organic matter content of soils.

The organic matter, clay minerals and sesquioxides represent the active fraction of soil, and silt and sand the inactive fraction. The active fraction, which is involved in chemical reactions in the soil, is colloidal in nature, i.e. the constituents representing this fraction are organic and inorganic substances with a small particle size (< 0.001 mm) rendering a large surface area per unit mass. According to Weber (1991a), it can be expected that the tilled surface soil is a relatively homogeneous mixture of the active and inactive fractions.

7.1.4 Organic matter

Organic matter is more flexible than clays with respect to the ability to adsorb (retain) pesticides. Well-decomposed organic matter has functional groups and ring structures that may carry either positive, negative or no charge, thus allowing the binding of both anions and cations of pesticides on a single organic matter molecule. Some degree of solubility is required for pesticides to become retained at the highly hydrated hydrophilic surfaces ((-)charge) of clay particles, whereas organic matter has both hydrophilic ("water-loving") and lipophilic ("fat-loving") characteristics which enable it to react with pesticides of wide ranging solubility and ionizability. According to Kononova (1966), and Weber, Swain, Strek and Sartori (1986), humified substances (humus) make up 17 to 97% of the total organic carbon in a soil. Humus consists predominantly of humic acids which are high molecular mass compounds with functional groups and aromatic rings which are lipophilic in nature and which possess numerous ionizable carboxyl and hydroxyl groups that give the polymer pH-dependent exchange properties (Kononova, 1966).

7.1.5 Clay minerals

Bohn, McNeal and O'Connor (1985), Koskinen and Harper (1990) and Weber (1991a, b) list some important characteristics of certain clay minerals which adsorb pesticides. The two major types of clay minerals present in soils are the 1:1 and 2:1 (silica:alumina) types (Brady, 1974). Kaolinite (1:1 clay mineral) does not exhibit high-intensity colloidal properties because of its limited adsorptive capacity for cations and a relatively low surface area. The intensive study of Frissel (1961) focused on the important role of montmorillonite and to a lesser extent illite and kaolinite in the adsorption of triazines in soil. Talbert and Fletchall (1965) reported that kaolinite at pH 5 and pH 7 did not adsorb either atrazine or simazine. In the 2:1 expanding type of clay, montmorillonite, there is a very large internal surface which far exceeds its external surface area. The combined internal and external surfaces (specific surface area) of montmorillonite greatly exceed the surface area of kaolinite. Negative charge predominates on the clay mineral surfaces, thus affording many adsorption sites for positively charged ions (cations) of pesticides.

7.1.6 Oxides and hydrous oxides of iron and aluminium

These colloidal fractions are intermixed with silicate clays in temperate regions. They are prominent in tropical and sub-tropical soils in several mineralogical forms, including boehmite, gibbsite, goethite and hematite (Brady, 1974). These minerals have a pH-dependent charge, and therefore may exist as positive, neutral, or negatively charged particles. The hydrous oxides have a much smaller cation exchange capacity (CEC) than kaolinite. In South Africa the amorphous Fe-Al-OH component is commonly found in acid soils in which kaolinite predominates. The Fe subfraction tends to be bonded strongly to crystalline minerals such as kaolinite, whilst the more weakly bonded Al fraction carries an active positive charge (Fouché and Brandt, 1973). In acidic soil, anions of pesticides will be attracted to and retained at the positive charges on the Fe-Al-OH fraction.

7.1.7 pH-dependent charge on soil colloids

There are two general sources of CEC for inorganic colloids: (a) permanent negative charge that results from isomorphic substitution, and (b) pH dependent charge which results from the ionization of hydrogens from hydroxyl (OH) groups located at the edges of micelles and from iron and aluminium hydroxides (Anderson, 1983). The greater surface area of the expanding type clays, combined with a much greater CEC, endows them with a much greater propensity for binding pesticides than the non-expanding clay types (Weber, 1972; Weber, Shea and Weed, 1986). Organic colloids will carry mainly negative charge at high soil pH, whilst positive charge will develop under acidic conditions. Pesticide

molecules that do not ionize in water are also retained by organic colloids, and therefore, this colloidal fraction has the capacity to bind pesticides that occur in the anion, cation or neutral form.

7.1.8 Influence of soil pH, water content, electrolyte concentration, temperature and pesticide "age" on adsorption

The primary roles of the physico-chemical properties of both the pesticide and the soil colloids in the adsorption-desorption phenomenon have been discussed above, but the availability of, e.g. herbicides for absorption by plant roots is known to be directly or indirectly influenced by several other factors. Although identification of the individual effects of soil factors involved with pesticide availability for uptake is often complicated by interrelationships between them, several factors which affect pesticide adsorption have been documented.

7.1.9 Soil pH

As alluded to earlier, soil pH influences pesticide adsorption through its effect on the charge characteristics of both the chemicals and soil adsorption sites. Weber and Whitacre (1982) found that atrazine is a weak base in aqueous solution and exists as a molecular species (zero charge) at high pH, and cations at low pH. Only atrazine in the cation form would be subject to adsorption on soil colloids with negative charge, thus explaining its increased adsorption and lower leaching/bioactivity at low pH in most soils. It was shown by Colbert, Volk & Appleby (1975) that adsorption of atrazine and terbutryn decreased on natural and limed soils as the soil pH increased to pH 8. Best and Weber (1974) found that the total amount of atrazine and prometryn applied was absorbed by plants over a five-month period ranged from 0.6 to 4.3% and was closely linked to the pH of the soil, with higher herbicide concentrations occurring in the plants at the higher soil pH level (7.7) compared to the lower pH (5.5). Consistent with the reported decrease in *s*-triazine adsorption with increased pH, Smit *et al.* (1979, 1980) showed that the phytotoxicity of atrazine in some soils was increased by increases in the soil pH from below pH 5 to about pH 6. They attributed this effect to increased availability of atrazine for uptake by plants, and also to increased chemical stability of atrazine molecules at the higher pH levels.

Several studies revealed that adsorption of *s*-triazines increased with a concomitant decrease in soil pH (McGlamery and Slife, 1965; Talbert and Fletchall, 1965; Weber, 1970b; Yamane and Green, 1972; Marshall, Nel and Smit, 1982). Protonation of atrazine molecules, and subsequent adsorption at negatively charged sites on soil colloids, progressively increases as soil pH decreases. Weber (1970b) suggested that atrazine may be bonded by complexation with protons on clay surfaces under acid conditions. Maximum adsorption of atrazine may be expected to occur near the pKa value (1.68) for this herbicide (McGlamery and Slife, 1965; Weber, 1970b). Reduced sorption is possible at pH levels lower than the pKa value, because of increased competition between hydronium ions (H_3O^+) and atrazine (cations⁺) for negatively charged sites on soil colloids.

Harris and Hurlé (1979) showed that atrazine and simazine adsorption onto clay colloids are highly sensitive to minute changes of soil solution pH and that plants can play an important role here by bringing about such changes in pH through ion exchange. They theorized that since triazine adsorption is so highly sensitive to pH changes, any change in rhizosphere pH through ion exchange or microbial activity must affect the adsorbed/solution herbicide equilibrium and, therefore, the amount of herbicide immediately available to the plant.

7.1.10 Soil water

The availability of pesticides for uptake by underground plant parts is influenced directly by soil water content, since water is the medium in which chemicals are transported in soil, and from which it is

either adsorbed or absorbed. Green and Obien (1969), Bailey and White (1970) and Appleby (1985) found an inverse relationship between atrazine adsorption and soil water content. Dao and Lavy (1978) suggested that soil water depletion would increase the atrazine concentration in soil solution, allowing polar atrazine molecules to compete favourably with a reduced number of water molecules for binding sites on soil colloids. Green and Obien (1969) were of the opinion that atrazine mobility in soil, and not fluctuations in atrazine concentration in soil solution *per se*, governed herbicide availability at the root surface. Several studies indicated that atrazine phytotoxicity was linked to soil water content, with increases in bioactivity as soil water content increased (Lavy, 1968; Dao and Lavy, 1978; Nel and Reinhardt, 1984).

The depth to which a pesticide leaches in soil is a function of its mobility, persistence, weather factors and percolating water. These parameters need to be considered in estimations of the potential of a pesticide to contaminate groundwater. Leaching of a compound beyond the root zone is dependent on the amount, frequency and intensity of water received. Apart from temperature, which has a significant effect on microbial activity and hence the persistence of compounds, the most important climatic characteristics relating to pesticide leaching are the duration of the "field capacity period" and the amount of "excess rain" during this period (Koskinen and Harper, 1990; Hollis, 1991). According to Talbert and Fietchall (1965) and Nel (1975), most of the atrazine applied at normal rates can be leached out of the root zone, but those fractions adsorbed on the internal surfaces of 2:1 swelling type clays, such as montmorillonite, would be least subject to leaching. Based on K_{oc} values, atrazine ($K_{oc} = 102$ to 163) is classified as moderately to slightly mobile (Leonard *et al.*, 1988). Because of the pH-dependent adsorption of atrazine on soil colloids, the herbicide is more susceptible to leaching under neutral and alkaline than under acid conditions (McGlamery and Slife, 1965; Smit and Nel, 1977).

Coarse-textured soils, which are low in organic matter and clay, would be particularly susceptible to pesticide leaching because these soils have limited sorption capacity. Fleming, Wax and Simmons (1992) contend that continued registration of atrazine for use in these soils may depend on development of formulation or application methods that reduce leaching potential. Controlled release formulations may have promise for reducing the mobility of atrazine. Fleming *et al* (1992) showed that a starch-encapsulated atrazine formulation (granules 20-40 mesh) should result in relatively fast atrazine release (compared to release from large granules of 14-20 mesh) and improved weed control (compared to weed control provided by the large granules and a commercial dry flowable formulation), while still reducing the leaching potential of atrazine.

According to Spencer and Cliath (1973), herbicides in the unsaturated soil solution may undergo some net upward movement with water as it moves from high to low potential via evaporation from the soil surface. Hubbs and Lavy (1990) states that the magnitude of this effect is proportional to the adsorbed and soil solution amounts of a compound. These researchers found that evaporation of water from the soil surface, and consequent upward movement of atrazine with capillary water, may be important in the dissipation of the herbicide.

Water solubility is often regarded as an indicator of adsorption and leaching. Retention of urea and triazine herbicides in the upper layers of the soil profile was originally attributed to their low water solubility. As discussed by Hartley (1976), it was subsequently recognized that adsorption by soil, not insolubility, was the most important factor retarding the movement of chemicals through soil. According to Briggs (1984), there is an inverse relationship between solubility and K_{ow} and hence K_d for liquids, whilst for solids an additional factor is the energy required to disrupt the crystal structure. This is related to the melting point (T_m) of the chemical (Briggs, 1981a) by the equation:

$$\log \text{water solubility (mole ml}^{-1}\text{)} = -\log K_{ow} - 0.01(T_m - 25)$$

It can be calculated that there can be a 1000-fold variation in water solubility between a liquid and a high melting point solid of the same K_{ow} and a corresponding 36-fold difference in K_d for compounds of the same water solubility (Briggs, 1984). Nicholls, Briggs and Evans (1984) showed that there are few chemicals for which water solubility is an important feature of their behaviour in soil. After an initial equilibration period all soil-applied herbicides are either adsorbed or dissolved at normal field rates, water solubility being a factor only for simazine and lenacil, each with $\log K_{ow} < 2$ and a high melting point (Briggs, 1984). The $\log K_{ow}$ values/melting points for atrazine and simazine are 2.0-2.5/173°C and 1.5-2.0/225°C, respectively. From equation (4), the water solubility of atrazine (33 mg L^{-1} at 25°C) would be expected to be higher than that for simazine (3.5 mg L^{-1} at 20°C).

Weber (1991a) states that organic matter is more universal in its ability to react with a wide assortment of herbicides with varying degrees of solubility and ionizability. Grover (1965) contends that under high soil water conditions, the availability of atrazine will predominantly be determined by the extent of its adsorption on the hydrophobic adsorptive sites on organic colloids. Under high soil water conditions adsorption on hydrophilic sites could be insignificant because atrazine would be readily desorbed from these sites by the overwhelming number of highly polar water molecules. Under low soil water conditions the availability of hydrophilic adsorptive sites will affect the bioactivity of atrazine. As the water content in the soil is decreased, more atrazine will be adsorbed on the now accessible hydrophilic surfaces (Grover, 1965).

7.1.11 Electrolyte concentration in soil solution

The differential availability of nutrients and soil-applied pesticides for uptake by plants depends on differences in their solubility in the soil solution and the extent and strength of adsorption. Hurle and Freed (1972) found that atrazine and simazine adsorption increased in the presence of rather high electrolyte (NH_4^+ , K^+ , Ca^{++}) concentrations and ascribed this effect to depressed herbicide solubility. Since the replacing power of cations on clay and humus in general follows the order $\text{C}^+ < \text{C}^{++} < \text{C}^{+++}$ (Brady, 1974), the divalent Ca^{++} ions would cover more of the negative adsorption sites and were more strongly adsorbed than the monovalent K^+ and NH_4^+ ions (Hurle and Freed, 1972). Herbicide adsorption could, therefore, be limited by competition between atrazine and cations for negatively charged adsorption sites, thereby increasing the concentration of atrazine in the soil solution. These findings were corroborated when Dao and Lavy (1978) showed that atrazine adsorption increased with an increase in the concentration of KCl and NH_4Cl in the soil solution. However, the electrolytic effect of CaCl_2 was comparatively low, probably because of competition between Ca^{++} and atrazine for adsorption sites (Dao and Lavy, 1978). Hurle and Freed (1972) are of the opinion that the effect of electrolyte concentration is negated under field conditions by constantly changing uptake by plants, dilution or precipitation.

7.1.12 Temperature

Temperature affects pesticide activity in various ways, often interrelated with other environmental factors. Temperature affects the bioactivity of pesticides by influencing their adsorption in soil, the rates of absorption by organisms, and their fate in those organisms.

Temperature is believed to exert an indirect influence on the adsorption process through its effect on pesticide solubility. Bailey and White (1970) reported the existence of an inverse relationship between the degree of adsorption and solubility within a number of herbicide groups, in particular *s*-triazines. McGlamery and Sife (1965) suggested that solubility play a role in the temperature effect on desorption as the solubility of atrazine is only 22 mg L^{-1} at 0°C, while it is 70 and 320 mg L^{-1} at 27 and 85°C, respectively.

7.1.13 Age of pesticides

Another factor affecting the bio-availability and dissipation of pesticides from soil is the length of time they have already been in the soil (Lavy, Mattice, Massey, Skulman, Senseman, Gbur and Barrett, 1996). Several studies have shown that differential dissipation rates exist soon after pesticide application (across days or months) and after the pesticide has aged (across months or years). All of these results are consistent with the compounds becoming less available with time due to diffusion into soil micropores or into or onto soil organic matter. Compounds showing increased adsorption are less available for uptake by plants, as well as for leaching to deeper depths.

7.1.14 Preferential flow in soil macro-pores

The discussion in previous sections centred on matrix flow of water and solutes. It would be incorrect to assume that knowledge of the roles of the factors discussed above will make the prediction of water and solute transport easy and accurate. Bypass flow in soil macropores may make prediction of leaching tenuous under certain circumstances. In contrast to matrix flow in micropores, relatively little adsorption of pesticides occurs in macropores and, as a result of intense rainfall or over-irrigation, transport of even normally highly adsorbed compounds may occur beyond the root zone. The term preferential flow refers to a range of physical non-equilibrium flow processes in soil. The process dominates on fine textured soil and occurs through macropores such as shrinkage cracks, worm holes and root pores. Soil macropores operate structures as high-conductivity flow paths, and by-pass the denser soil matrix (Jarvis, 1996). Preferential flow can also occur in unstructured sandy soil as a result of profile heterogeneity such as at horizon interfaces, due to texture variations and even through water repellence. Preferential flow will also occur on homogenous soils where sub-surface drainage is prevalent (Van der Zee and Boesten, 1991). During preferential flow, solutes leach more rapidly than would be expected. This is because the solutes in the fast flowing water do not have sufficient time to equilibrate with the slow moving or stagnant water contained in the bulk of soil matrix. Preferential flow is important in pesticide leaching because the pesticide rapidly by-passes the top-soil, into the sub-soils. In the sub-soils, adsorption and degradation generally greatly decreases or slows down. From the available experimental data it can be concluded that preferential flow is rather the rule than the exception (Flurry *et al*, 1994). This means that the models that do not take these processes into account may not be that widely applicable. A number of models have been developed which take preferential flow into account. There are two major groups of preferential flow models, those, which use microscopic approaches to calculate transport at the pore, scale and those that follow the macroscopic continuum approach. The continuum approach divides the soil into flow domains, each characterised by a water flow rate and solute concentration. This means that additional parameters are required for the multi-domain models. These include fractions of sorption sites per domain, the diffusion path length which regulates exchange between domains and soil hydraulic conductivity defining each domain.

The presence of preferential pathways in soil may explain fast vertical movement of pesticides. Therefore, although the soil constituents are responsible for retaining and regulating herbicide mobility and bio-availability, the structure and architecture of the soil also play critical roles. Soils containing channels from earthworm activity, dead roots or voids from expansion and contraction normally have high infiltration rates and water may move through these macropores and cracks faster and in greater amounts than through soils with undisturbed matrixes. Pesticides dissolved in void-occurring water should move faster and to greater depths than water in smaller openings (e.g. soil micropores), providing the chemicals were dissolved in the water phase and were not mixed with the soil media (Koskinen and Harper, 1990; Weber, 1991a, b; Aderhold and Nordmeyer, 1995). Fortunately, the latter is normally the case, and therefore, the flow of pesticides is usually impeded, to some extent at least.

7.1.15 Volatilisation

In addition to sorption, the distribution of the solute between the gas and liquid phase will also influence leaching. This is a function of the volatility of the compound being investigated. Volatilisation is the process in which a chemical evaporates in the vapour phase to the atmosphere from a different environmental compartment. The rate of evaporation is dependent on chemical properties of the compound, as well as environmental conditions. Where pesticides are applied directly to soil it is assumed that loss through volatilisation is negligible, and is thus not taken into account. The situation changes where pesticides such as fumigants are considered.

Pesticides may move from one compartment of the environment to another in vapour form. The propensity of pesticides to convert from the solid or liquid phase to the gaseous state is strongly linked to a compound's vapour pressure. Generally, the higher the vapour pressure of a compound the greater the chance that losses through volatilisation will occur. Incorporation or leaching of pesticides into soil normally reduces volatility loss. Compounds that are strongly adsorbed on soil colloids are less likely to vaporise compared to those that are weakly held or occur mainly in the water phase of the soil. In anticipation of volatility loss, volatile compounds are either injected into the soil or incorporated mechanically immediately after application.

7.1.16 Transformation Processes

The above discussion of transfer processes involved in the determination of pesticide behaviour has dealt with some important components of the complex of factors which governs their mobility, bioactivity and persistence in soil. Although degradation of pesticides in soil is only one component of this complex, it is generally considered to be the principal route for loss of most compounds from soil (Wolfe, Mingeir and Miller, 1990; Bollag and Liu, 1990; Walker, 1987, 1989). Soils provide an ideal environment for many types of degradation processes. Soil is a dynamic and complex biological and chemical medium in which numerous variables interact to determine the degradation rate of a compound and, hence, its persistence. In this context, factors such as soil water content, temperature, soil texture, nutrient status, organic matter, pH and microbial activity are regarded as important variables. Degradation processes usually involve direct chemical transformation of the compound, and the activities of soil micro-organisms.

7.1.17 Soil properties

Organic matter content of soil could be of particular importance in determining the degradation rate and thus the persistence of pesticides. The dominant role of organic matter in the adsorption of pesticides was cited in the above discussion. Organic matter also regulates microbial activity of soil - with generally higher levels of biomass and respiration in more organic soils (Walker, 1989). The clay content of soil may also be important in determining pesticide persistence, because as pointed out earlier, it is an important predictor of a compound's availability in the soil solution. When working with natural soils, it can be difficult to separate the effects of clay and organic matter since they are often correlated (Eagle, 1983a; Reinhardt and Neil, 1989). Clay content is an important component of soil texture, and soil texture can have a marked effect on other soil properties. Coarse-textured soils with low clay content tend to have less organic matter and lower microbial biomass than fine-textured soils (Walker, 1989). Soil texture will also influence pore size distribution in soil and consequently water relations and aeration characteristics.

Soil pH can have a pronounced effect on pesticide degradation rates. As pointed out earlier, pH exerts a direct effect when the stability of the chemical is pH-dependent, or it has an indirect effect through changes in microbial populations, or through colloidal adsorption of the compound.

7.1.18 Temperature and water

In addition to the variability in pesticide degradation rates between soils, rates of loss may vary in a particular soil according to the water and temperature regimes encountered. In general, rates of pesticide loss increase with increases in temperature and soil water (Walker, 1989). Increased atrazine degradation rates with increasing temperature and soil water have been demonstrated in field experiments (Harris, Woolson and Hummer, 1969) and in laboratory incubation studies (Walker and Zimdahl, 1981). With many pesticides, Briggs (1983) and Walker and Allen (1984) demonstrated a 2 to 2.5 fold increase in half-life with a 10°C decrease in temperature, and a 1.5 to 2.5 fold increase in half-life if soil water is reduced by a factor of two. Weather conditions after herbicide application in the field should have a similar large effect on persistence. The weather, therefore, can have a marked influence on rates of pesticide loss from soil, and it is essential to be aware of this when evaluating field persistence data from different sites.

Duffy (1991) states that in warmer climates (or in years with above normal temperatures) degradation by chemical reactions will typically be faster than that observed in cooler climates (or years having above normal rainfall). This relationship also holds for microbial degradation processes, at least until temperatures are so high that death or dormancy in microbial populations occurs.

In drier climates (or in years having below normal rainfall) the probability that a herbicide will remain in the root zone is usually higher than in wetter climates (or in years having above normal rainfall). This is due to the lower net precipitation of the drier climate or year and/or the greater upward movement of soil water which would move at least a fraction of any pesticide that had been leached deeper into the profile back into the root zone (Duffy, 1991).

7.1.19 Chemical degradation

Degradation of herbicides in soils by purely chemical reactions has been discussed by Wolfe *et al.* (1990) and Kearney and Kaufman (1988), and the disposal of pesticide wastes has been discussed by Krueger and Seiber (1984). Chemical hydrolysis proceeds by hydrolysis, oxidation and reduction reactions.

Chemical hydrolysis of atrazine to hydroxyatrazine is regarded as the main avenue of inactivation of the herbicide in soil (Armstrong, Chesters and Harris, 1967; Gamble and Khan, 1985). The atrazine molecule is stable under neutral pH conditions, but rapid chemical hydrolysis occurred under highly acid or alkaline conditions (Armstrong *et al.*, 1967). In studies with simazine (Walker *et al.*, 1983) and metribuzin (Allen and Walker, 1983), degradation rates decreased as adsorption on soil colloids increased, presumably due to decreased availability for degradation. Adsorption, however, does not always lead to protection from degradation, and examples of adsorption-catalysed hydrolysis of some chloro-triazines, including atrazine, have been reported (Armstrong and Chesters, 1968; Hance, 1979). Armstrong *et al.* (1967) and Swain (1981) ascribed the major role of organic matter in the inactivation of atrazine in soil to its influence on the rate of atrazine hydrolysis. Organic matter evidently catalyses hydrolysis through adsorption of compounds on this soil constituent, and therefore, the rate of hydrolysis reactions is rapid in soils with a relatively high organic matter content compared to soil with low organic matter contents.

7.1.20 Biological degradation

This includes decomposition by microbes and metabolic decomposition by plants. Some microbes have the ability to use herbicides as a food source, expending O₂ in the process releasing CO₂. The higher the microbial activity of a soil, the higher the rate of microbial degradation to be expected for those pesticides prone to microbial degradation. Generally, the higher the soil organic matter content,

the higher the microbial activity. Soil microbial activity is also affected by temperature, water, O_2 , pH and nutrient availability. Thus if a herbicide is prone to microbial degradation, the residual activity of such a compound will also be affected by these factors.

Most organic pesticides are prone to microbial breakdown in soil (Bollag and Liu, 1990). Many micro-organisms are known to use organic chemicals as energy sources - they feed on them, and in the process degrade (inactivate) them. As atrazine is chemically stable at neutral pH values (Armstrong *et al.*, 1967), it is likely to persist in many soils and chalky groundwater (Wood *et al.*, 1991) unless degraded by micro-organisms. Kaufman and Kearney (1970) listed a large number of micro-organisms that have the ability to degrade atrazine in pure culture, most of those reported being fungi. There are, however, reports of bacteria including *Arthrobacter* sp., *Bacillus* sp. and *Pseudomonas* sp. (Kaufman and Kearney, 1970; Cain and Head, 1991). Oxidative dealkylation appears to be the major mechanism by which micro-organisms degrade atrazine, but degradation has also been observed under anaerobic conditions (Kaufman and Kearney, 1970).

7.1.21 Photo-decomposition

Excessive activation by sunlight of electrons in pesticide molecules may cause loss of structural integrity, which results in biologically inactive residues. This route of degradation is only important for pesticides that remain on the soil surface for a long time, or those that are exposed to sunlight in clear water. Mechanical incorporation of light-sensitive compounds in the soil is advised or else rain is required soon after application in order to leach the pesticide into the upper soil zone.

The length of time that a pesticide remains active in soil is a function of the dosage rate and the rate of loss from the soil. The rate of loss is determined by the combination of the degradation processes and the transfer processes. Persistence is thus a function of degradation. Degradation occurs through biological decomposition (microbes and plants), chemical degradation and/or photo-decomposition. Not all pesticides are prone to all three these degradation processes.

7.1.22 Pesticide Persistence

The above discussion has given an indication of how variations in some soil properties can influence rate of degradation of pesticides through effects on adsorption, chemical transformation, or microbial breakdown. The end result is variability in degradation rate, and therefore, persistence that varies from soil to soil and from site to site. Persistent soil-applied herbicides such as atrazine are an essential component of many crop production systems. The time that herbicides persist in the soil is of particular importance as this has already been shown to have serious implications for the safety of sensitive following crops (Eagle, 1978; Caverley, 1983; Gottesbüren, Pestemer, Wang, Wischnowsky and Zhao, 1991).

The persistence of a pesticide is mostly characterised by the half-life of the compound. The half-life or DT_{50} is defined as the time in days required for the concentration of a compound to be halved, under a given set of environmental conditions. Pesticide half-life is sensitive to environmental conditions such as soil temperature, soil water content, air temperature, ultra-violet radiation and microbial activity of the soil. It must be kept in mind that degradation may occur during any stage of transfer.

From an agronomic viewpoint, the ideal herbicide should persist long enough to provide season-long weed control but not so long that its residues harm sensitive follow-up crops. Carry-over problems are likely to occur when soil and weather factors which favour reduced herbicide degradation prevail periodically or for a significant length of time subsequent to herbicide application; and when the particular follow-up crop is markedly sensitive to the specific herbicide residue. Excessive persistence tends to occur more frequently in some seasons than in others (Eagle, 1983b), reflecting the influence

of climatic factors on rates of loss. Some soils are also more prone to carry-over problems than others. These problems are exacerbated by errors in initial application which lead to overdosing.

The persistence of pesticides is the overriding factor in the determination of their potential to reach groundwater sources in detectable amounts. For example, a highly leachable compound that has a very short persistence is unlikely to pose a threat to groundwater, provided the water table is not very high. Even if such a compound does reach the groundwater, its limited persistence will probably render it harmless quickly. On the other hand, a highly persistent compound with low mobility in soil, may eventually accumulate to unacceptable levels in groundwater if given enough time. Thus, both persistence and adsorption/leaching characteristics of pesticides must be considered in assessments of their potential to contaminate groundwater. No pesticide has an inherent persistence (expressed as half-life, i.e. the time required for the original concentration to be halved in a particular situation). In earlier discussion it was pointed out that, due to the influences of many factors and their interactions, the persistence of a compound can vary considerably from soil to soil, from one site to another, from year to year, and from season to season. Leaching behaviour can also not be expected to be constant for a particular compound. For these reasons computer models that predict pesticide leaching and persistence are extremely helpful tools in pesticide behavioural studies. Prerequisites for accurate predictions include the availability of relevant input data and validation of models in the field.

7.2 MODEL DEVELOPMENTS AND CURRENT STATUS

Modelling became, in the past few years, an effective tool for decision-making and management related to the environmental impact of pesticides applied on agricultural land. In many countries, models are also used to determine the hazard of new pesticide products for registration purposes (Boesten, 1999; Vanclooster *et al.*, 2000a).

In order to model the behaviour and fate of pesticides, it is essential to identify the processes that occur once these compounds are applied on agricultural land. The major processes identified and dealt with in this study on modelling are summarized in Table 7.1. These can be grouped into transfer and transformation processes. The factors that influence these processes and mechanisms are also discussed in this study. The main factors involved are:

- Nature of the pesticide (reactivity, solubility, age etc.)
- Management (time of application, amounts, application of soil amendments and tillage, etc.)
- Environmental factors (weather conditions, soil properties etc.)

A number of minor processes occurring in nature were also identified. These are mentioned in this study, but they will be not dealt with in modelling because their effect on the total balance is expected to be negligible.

Table 7.1 Major processes occurring at field scale for modelling the behaviour and fate of pesticides

Processes	Sub-processes
Transfer	Runoff Volatilisation Adsorption/desorption Plant uptake Soil fluxes (water and solute fluxes, leaching, preferential flow and intermediate vadose zone)
Transformation	Chemical degradation Microbial degradation Photo-decomposition

Comprehensive reviews of pesticide models were recently published by Boesten (1999) and Vanclouster *et al.* (2000a). A review of models currently in use world-wide and their main features are shown in Table 7.2.

Table 7.2 Review of models currently in use and their main features (after Vanclouster *et al.*, 2000a)

Model	Features					Author
	Water flow	Solute transport	Preferential flow	Sorption	Volatilization	
CRACK-NP	Capacity	Convection	Yes	Linear	No	Armstrong <i>et al.</i> (2000)
GLEAMS	Capacity	Convection	No	Linear	No	Rekolainen <i>et al.</i> (2000)
LEACHP	Richards	Convection-dispersion	Yes	Freundlich	Yes	Hutson and Wagenet (1992); Dust <i>et al.</i> (2000)
MACRO	Richards	Convection-dispersion	Yes	Linear	No	Jarvis <i>et al.</i> (2000)
PELMO	Capacity	Convection	No	Freundlich	No	Klein <i>et al.</i> (2000)
PESTAN	-	Convection	-	-	-	Cohen <i>et al.</i> (1995)
PESTLA	Richards	Convection-dispersion	No	Freundlich	No	Boesten and Gottesburen (2000)
PESTRAS	Richards	Convection-dispersion	No	Freundlich	Yes	Freijer <i>et al.</i> (1996)
PLM	Capacity	Convection	Yes	Linear	No	Nicholls <i>et al.</i> (2000)
POLLUTE	-	-	-	-	-	Rose and Ghadiri (1992)
PRZM2	Capacity	Convection	No	Linear	No	Trevisan <i>et al.</i> (2000a)
SESOIL	-	-	-	-	-	Bonazountas (1983)
SIMULAT	Richards	Convection-dispersion	Yes	Freundlich	No	Aden and Dieckkruger (2000)
VARLEACH	Capacity	Convection	No	Linear	No	Trevisan <i>et al.</i> (2000b)
WAVE	Richards	Convection-dispersion	Yes	Linear	No	Vanclouster <i>et al.</i> (2000b)

7.2.1 Transfer Processes

Runoff

Along with leaching, runoff is the most direct process involved in the contamination of water resources. In runoff water, pesticides are transported as both solutes and adsorbed on suspended particles (Rose and Ghadiri, 1992). Gouy *et al.* (1999) determined sorption isotherms on both soil and eroded particles and indicated that the percentage of pesticide transported through the liquid or adsorbed phase varies greatly depending on the pesticide species. In order to determine the solute component, soluble pesticide runoff can be calculated as a function of rainfall or irrigation. The soluble pesticide runoff concentrations are generally assumed to be proportional to the pesticide concentration in the top centimeter of the soil (Cohen *et al.*, 1995). Adsorbed pesticides need to be associated with sediment transport processes (Johanson, 1983). It is also important to quantify the organic matter coating suspended particles as this could considerably increase sorption of pesticides (Zhou *et al.*, 1995).

Several models have been developed to predict pesticide runoff (Williams, 1998a and b; Inao and Kitamura, 1999). Most models use the semi-empirical method based on the USDA SCS runoff curve number (USDA SCS, 1972) to estimate water runoff (Ma *et al.*, 1998). For example, the GLEAMS model (Rekolainen *et al.*, 2000) uses the USDA SCS runoff curve number (USDA SCS, 1972) to estimate water runoff, whilst it estimates sediment loss using the Universal Soil Loss Equation (USLE) (Wischmeier, 1978). The quantity of pesticide runoff is then calculated with:

$$C_{av} = C_w / B + C_s$$

C_{av} – Runoff-available concentration of pesticide
 C_w – Concentration of pesticide in water
 B – Extraction coefficient
 C_s – Concentration of pesticide in soil

The extraction coefficient B is modeled as a function of the adsorption coefficient K_d , which varies from 0.1 to 0.5 (Rekolainen *et al.*, 2000). The GLEAMS model assumes that the top 10 mm of the soil supply effectively pesticide for runoff.

The pesticide removal from a unit area of land (N) through transport on suspended, eroded soil particles, can also be described with a more empirical function (Yaron *et al.*, 1996):

$$N = ER M C_s$$

ER = g chemical g⁻¹ soil eroded sediment / g chemical g⁻¹ original soil
 M – Accumulated mass of chemical per unit area (g chemical m⁻²)

Volatilisation

Volatilisation is the process where solutes move from the medium where they are dissolved into the atmosphere. It can occur from plant and soil, but this process is generally not well understood (Boesten, 1999). Volatilisation from the plant surface is most likely a function of saturated vapour pressure (Woodrow *et al.*, 1997; Smit *et al.*, 1998). Volatilisation from the soil can occur from the very thin surface film after spraying and before rainfall and/or irrigation, as well as, in the case of volatile pesticides, from the soil matrix via vapour fluxes (Boesten, 1999). The top soil layer from which volatilisation occurs is generally in the order of 5 mm (Jury *et al.*, 1983; Hutson and Wagenet, 1992). A rough simplification can be made by correcting the application rate for the expected loss, as most losses in the field occur within one week after application (Vandoooster *et al.*, 2000a). The process of volatilisation can be associated with losses of pesticide during spraying. Vandoooster *et al.* (2000a) reviewed models that include volatilisation.

Volatilisation can be modelled adopting Henry's law, used in LEACHP (Hutson and Wagenet, 1992):

$$C_g = H C_w$$

C_g – Concentration of pollutant in the gas phase
 H – Henry's constant

Henry's constant is typical for a pesticide species. Chemicals with low Henry's constant are more prone to accumulate and increase at the soil surface. This process can then be modelled as a function of the air-boundary layer thickness and water evaporation rate. Accumulation of pesticides at the soil surface can also increase photo-decomposition and runoff into surface waters, as well as change adsorption coefficients in the top soil layer.

In terms of flux, volatilisation can be expressed as (Hutson and Wagenet, 1992; Klein *et al.*, 2000):

$$J_v = -D H C_w / d$$

J_v - Volatilisation rate

D - Diffusion coefficient in air

d - Top soil layer from which volatilisation occurs

In LEACHP, the diffusion coefficient can be modified to account for changes in soil water content, barometric pressure and temperature that may cause air flow through soil (Scotter *et al.*, 1967). More detailed models include losses by volatilisation in the convection-dispersion equation, which is used to describe solute fluxes in the soil profile (Hantush *et al.*, 2000).

Adsorption/desorption

Adsorption is the adhesion of pesticide molecules to soil solid particles. This process influences the pesticide fluxes by retarding or enhancing pesticide migration.

Bonazountas (1983) proposed a very simple model for cation exchange:

$$S = \text{CEC MWT} / \text{VAL}$$

S - Maximum mass of pesticide associated with solid

CEC - Cation exchange capacity

MWT - Molecular weight of pollutant

VAL - Valence of ion

Considering that the largest component of pesticide adsorption is generally associated with organic matter, the equilibrium partitioning between soil water and solids can be simply written as (Bomberger *et al.*, 1983):

$$C_s = K_{oc} f_{oc} C_w$$

K_{oc} - Organic carbon partition coefficient

f_{oc} - Fraction of the soil solid phase that is organic

Besides these simplified methods, adsorption/desorption can be quantified with linear or non-linear sorption isotherms (Pepper *et al.*, 1996):

$$\text{Linear isotherm: } S = K_d C_w$$

K_d can be calculated as (Dust *et al.*, 2000):

$$K_d = K_{oc} f_{oc}$$

The linear sorption is used, for example, in LEACHP (Hutson and Wagenet, 1992).

$$\text{Non-linear (Freundlich) isotherm: } S = K_f C_w^n$$

where n is a power function related to the sorption mechanism and K_f can be defined as the Freundlich isotherm.

Vanclooster *et al.* (2000a) reviewed models using either linear or non-linear sorption (Table 2). Linear or non-linear sorption can be included in the convection-dispersion equation. Alternatively, an approach using selectivity coefficients could be used to determine adsorption/desorption, but this would require more inputs. The adsorption/desorption processes are generally assumed to be instantaneous. Their dependence on time can be modeled if the kinetic process is known, but this would also require more inputs. The effect of hysteresis on adsorption/desorption was also proven and should be taken into account (Leake and Gatzweiler, 1995).

Plant uptake

Uptake of pesticides by plants is generally modeled based on the water uptake and the concentration of pesticides in the soil solution to follow a linear model (Hutson and Wagenet, 1992; Klein *et al.*, 2000):

$$J_{UP} = F_{UP} C_w T$$

J_{UP} – Uptake flux

F_{UP} – Plant uptake factor

T – Transpiration flux

In the WAVE model (Vandclooster *et al.*, 2000b), a mechanistic calculation of transpiration is introduced as a function of potential transpiration, root density distribution and soil matric potential. Exudation and retention by plants can also be modelled, but these processes are not deemed to have a major effect on the total balance.

Soil fluxes

Water and solute fluxes

A reliable prediction of water fluxes is the first condition for modelling pesticide transfer in the soil profile. The second condition is accurate simulation of pesticide fluxes. Heat fluxes required to predict, for example, pesticide solubility and degradation are not deemed to be relevant, as the uncertainty of soil temperature estimation is higher than the uncertainty of its impact on pesticide degradation (Vandclooster *et al.*, 2000a).

Water redistribution in the soil profile can be modelled using tipping bucket (cascading) models based on soil-specific field capacity levels, or finite difference models based on the solution of Richards' continuity equation. Solute redistribution can be modelled assuming complete mixing, piston flow, convection or convection-dispersion. Vandclooster *et al.* (2000a) reviewed models solving water and solute redistribution adopting different approaches (Table 7.2). Similarly, Pepper *et al.* (1996) classified the modes of modelling pesticide transport and leaching into:

- Convection or mass transport (passive movement of chemicals through the soil with water).
- Convection-dispersion flow (movement by convection, dispersion due to variations in velocity through pores of different size, and diffusion governed by Fick's law based on differences in concentration gradients over limited distances).

The state-of-the-art approach for simulating water and solute fluxes in soils makes use of a finite difference solution to Richards' continuity equation. For example, Pang and Letey (1999) developed a pesticide transport module in the convection-dispersion equation for solutes to predict pesticide movement through soils and leaching, accounting at the same time for adsorption and biochemical decay of the pesticide:

$$\partial C_p / \partial t = D_R \partial^2 C_p / \partial z^2 - V_R \partial C_p / \partial z - \mu C_p$$

C_p - Pesticide concentration

$D_R = D_p / R$

D_p - Combined diffusion and hydrodynamic dispersion coefficient

R - Retardation factor

$V_R = v / R$

v - Pore water velocity

$R = 1 + \rho_b K_d / \theta$

ρ_b - Soil bulk density

θ - Volumetric soil water content

K_d - Distribution coefficient equal to the slope of linear adsorption isotherm

$$\mu = \ln(2) / T_{1/2}$$

μ - First-order decay rate constant of pesticide

$T_{1/2}$ - Half-life of pesticide

R - Gas constant

In their work, Pang and Letey (1999) also investigated the effect of different factors (nitrogen applications, irrigation management and soil salinity) on pesticide transport and leaching. This model was expanded by Hantush *et al* (2000) by replacing D_R with the effective liquid phase diffusion coefficient and μ with the effective first-order loss rate coefficient to account for volatilisation and crop uptake.

The convection-dispersion approach has a more scientifically sound background, but it requires more inputs compared to the convection approach, in particular if processes like volatilisation, crop uptake and biochemical decay are solved simultaneously.

Leaching

Along with runoff, leaching is the most direct process involved in the contamination of water resources. Therefore, water and solute fluxes in the soil profile need to be accurately described in order to accurately predict leaching. Leaching is the amount of pesticide dissolved in the solution percolating through the bottom of the active root zone.

Preferential flow

Preferential (macropore or by-pass) flow needs to be simulated in the presence of cracking soils, but this process is generally not well understood (Boesten, 1999). Vanclooster *et al* (2000a) reviewed models accounting for preferential flow (Table 7.2).

CRACK-NP is a pesticide model used to predict leaching in cracking clay soils (Armstrong *et al*, 2000). Mass transport of solute (pesticides) is calculated from the amount of water flowing in the cracks between the layers and the solute concentration of the crack water in each layer. Solute diffusion between crack water and aggregate water is simulated. The shortcoming is that CRACK-NP can only be used in clay soils where the dominant water flow is through cracks and fissures as it assumes that water flow in soil matrix aggregates is negligible.

On the other hand, modelling preferential flow by using the mobile/immobile approach for water and solute flow in soils is becoming more and more popular (Cote *et al*, 1999). For example, PLM simulates preferential flow of water and pesticides by establishing a mobile and immobile category of soil solution, where the mobile category represents the macropores in structured soils (Nicholls *et al*, 2000). The mobile category is also divided into "slow" and "fast", where preferential flow occurs only when rainfall or irrigation intensity is so high to replenish the "fast" mobile phase. The model can then be used to predict preferential flow if the percentage of "fast" pores in the mobile phase is calibrated (input data). Caution should, however, be exercised in the use of this empirical method because the percentage of "fast" pores changes with location and time.

Hantush *et al* (2000) described a model combining the mobile/immobile phase with the convection-dispersion equation. SWAP employs the clay shrinkage characteristic, based on the relationship between the void ratio and the moisture ratio, for the simulation of soil crack formation and water and solute transport in the mobile and immobile phase (Huygen *et al*, 2000).

Besides the lack of understanding of preferential flow processes, models were also not sufficiently validated. The number of specific cases is too large and there is, therefore, need to develop and adapt existing models.

Intermediate (vadose) zone

The intermediate (vadose) zone is becoming more and more the topic of research investigating groundwater impact and vulnerability to contaminants. The rate of pesticide transfer and transformation in sub-soils is often unknown and difficult to establish (Vandclooster *et al.*, 2000a). It is different from the root zone because of lack of organic matter, different temperatures, microbial activity, water content, texture and structure. For example, it was proven that the accuracy of the PELMO (Klein *et al.*, 2000) and VARLEACH (Trevisan *et al.*, 2000b) models decreased by increasing depth (Borchers *et al.*, 1995). Besides this, the principles of pesticide transfer in the intermediate zone, including preferential flow, are similar to those applied in the root zone (Hantush *et al.*, 2000). There is need to extend the description of water and pesticide fluxes from the root zone to the intermediate zone.

Alternatively, models developed for vulnerability assessment of groundwater impact could be adapted and used in combination with soil water and solute flux models. The most commonly used model for this purpose is DRASTIC (Aller *et al.*, 1987). The DRASTIC method (Aller *et al.*, 1987), or its simplified DAS index, is a particularly powerful tool when it is applied at large scale. DRASTIC is a model that could be used for semi-qualitative studies on groundwater vulnerability for generic contaminants. It does not, however, describe specific pesticide-related processes.

Jury *et al.* (1987) derived a model for vulnerability assessment of pesticides:

$$M(z) / M_0 = AF$$

$M(z)$ – Mass of pesticide that reaches the water table

M_0 – Mass of pesticide applied to the soil surface

AF – Attenuation factor

The attenuation factor index (Rao *et al.*, 1985) can be used for pesticide vulnerability assessment:

$$AF = \exp[-0.693 Z R \theta_{FC} / q T_{1/2}]$$

Z – Depth to water table

θ_{FC} – Volumetric water content at field capacity

q – Average annual groundwater recharge

$$R = 1 + D_b f_{oc} K_{oc} / \theta_{FC} + \theta_g H \theta_{FC}$$

θ_g – Gas content

A leaching potential index (LPI) can also be used by inverting the exponential argument of AF (Meeks and Dean, 1990):

$$LPI = 1000 T_{1/2} V / (0.693 R Z)$$

V – Soil water velocity

Schlosser and McCray (2002) did a sensitivity analysis using LPI and concluded that organic carbon content, depth to groundwater and soil water velocity are the main factors affecting pesticide leaching, whilst the effect of bulk density and soil water content is less pronounced. Major factors are also the organic carbon partition coefficient and pesticide half-life. These are, however, dependent on pesticide type, site and environmental conditions.

Another possible option is to combine soil water and solute flux models extended to the intermediate vadose zone with groundwater models, like for example MODFLOW (McDonald and Harbaugh, 1988).

The approach to be used will depend on the purpose and objectives of this research.

7.3 TRANSFORMATION PROCESSES

7.3.1 Chemical degradation

The factors controlling the degradation rate of pesticides are not always well understood (Boesten, 1999) as degradation (persistence) is a complex process and it is not an inherent property of the pesticide. It is dependent on soil properties, weather conditions and properties of pesticide (reactivity, solubility, age, etc.). Aden and Diekkruger (2000) suggested several approaches to describe pesticide degradation in the SIMULAT model (O'Neill optimum curve, Michaelis-Menten kinetic, metabolic or co-metabolic degradation).

Chemical degradation and decomposition due to microbial activity and light (photodecomposition) are often expressed simultaneously by first order kinetics (Jury and Masoud, 1989; Cohen *et al.*, 1995):

$$C_{t_2} = C_{t_1} \exp[-K (t_2 - t_1)]$$

C_{t_2} – Concentration at time t_2

C_{t_1} – Concentration at time t_1

K – Degradation rate constant

Simple exponential decay can also be described in terms of field half-life $T_{1/2}$, generally expressed in days:

$$T_{1/2} = \ln(2)/K$$

The ideal pesticide and environmental conditions will induce enough activity for the pesticide to have the desired effect on pests, but enough short persistence to limit leaching and contamination of groundwater. In many instances, adsorption/desorption processes are simultaneously included with degradation to estimate leaching. For example, Cohen *et al.* (1995) used the following equation to estimate leaching:

$$AF = \exp(-B) = M_2/M_0$$

AF – Pesticide decay

M_2 – Mass lost below the root zone

M_0 – Amount of pesticide applied to the soil surface

$B = K(tr)$

tr – Pesticide travel in the vadose zone

$tr = Z RF \theta_{FC} / q$

RF – Retardation factor $(1 + \rho_b + K_d / \theta)$

Gustafson (1989) suggested the GUS (Groundwater Ubiquity Score) screening index as a function of half life ($T_{1/2}$) and the partitioning coefficient between the soil organic carbon content and the soil solution (K_{oc}):

$$GUS = \log(T_{1/2}) * [4 - \log(K_{oc})]$$

Empirical equations of the following form can also be derived for specific sites and conditions:

Potential leaching depth = $a + b * \%C + c * \text{pH of the soil}$

Potential leaching depth = $a + b * \%C + c * \text{rainfall/irrigation}$

7.3.2 Microbial degradation

The half life of pesticides, which is not an inherent property, will depend on microbial activity, which depends on temperature and soil moisture. It is therefore very important to indicate the conditions under which biochemical degradation occurs. The microbiological activity can be predicted depending on base, optimal and maximum temperatures (Monteith, 1977), and this will induce a temperature-degradation rate response function. The effect of soil water content on degradation of pesticides can be modelled with a similar response function (Vanclooster *et al.*, 2000a).

The response of degradation to temperature can be modelled using the Arrhenius function (Aden and Diekkruger, 2000):

$$K(T) = K_0 \exp(E_a / RT)$$

T - Actual ambient temperature
 E_a - Activation energy
 K_0 - Baseline degradation rate
 R - Gas constant

The soil temperature can be described as a function of air temperature (Campbell and Norman, 1998).

Jarvis (1994) expressed the effect of soil temperature on degradation through a factor, F_T :

$$F_T = \exp[\alpha(T - T_{ref})]$$

T_{ref} - Reference soil temperature
 α - Degradation rate coefficient

The influence of soil moisture can be calculated using an empirical factor b , with:

$$F_w = (\theta / \theta_s)^b$$

θ_s - Boundary volume fraction of water

Actual volumetric soil water content is calculated from the water balance.

LEACHM (Hutson and Wagenet, 1992) uses first-order transformation and degradation rate constants, which can be corrected for water content and temperature.

7.3.3 Photo-decomposition

Photo-decomposition is a photochemical reaction due to absorption of electromagnetic energy by a pollutant. It occurs at the plant and soil surface exposed to sunlight, before the first rainfall/irrigation event for many pesticides (Walker, 1974), and there is evidence that it can contribute considerably to the pesticide balance (Bavcon, 2003). However, due to the difficulties in micro-measurements at the soil surface, similarly to volatilisation, this process was not studied enough. Both photo-decomposition and volatilisation can be reduced or eliminated by shallow incorporation of the pesticide applied into the soil (Bromilow *et al.*, 1999).

7.4 DISCUSSION AND CONCLUSIONS

7.4.1 Review of modelling approaches

Based on the literature review, the following considerations on pesticide-related processes can be made:

- Pesticide runoff is modelled to occur in solution from the top centimeter of the soil and as adsorbed on solid mineral and organic particles. Most models use the semi-empirical USDA SCS runoff curve number method (USDA SCS, 1972) to estimate dissolved pesticides and sorption isotherms to determine adsorbed pesticides.
- Volatilisation from plant and soil surface can be modelled using Henry's constant, which is specific for the pesticide species.
- Adsorption/desorption of pesticides is modelled depending on pesticide species and soil properties, where the organic component is generally dominant compared to the mineral fraction. The fraction of pesticides adsorbed on soil solid particles is modelled using linear or non-linear (Freundlich) isotherms.
- Plant uptake of pesticides is modelled as a function of water uptake and concentration of pesticide in the soil solution.
- Water redistribution in the soil profile can be modelled using tipping bucket (cascading) models based on soil-specific field capacity levels, or finite difference models based on the solution of Richards' continuity equation. Solute redistribution in the soil profile and leaching can be modelled using the complete mixing, piston flow, convection or convection-dispersion approaches. Preferential (macropore or by-pass) flow can be modelled by differentiating between mobile and immobile phases.
- Chemical degradation is modelled as a function of pesticide half-life using first order kinetics.
- Soil water content and temperature are the driving forces for microbial degradation.
- Photo-decomposition can be modelled as a function of incoming solar energy and concentrations of pesticides on plant and soil surfaces.

In summary, state-of-the-art models use the convection-dispersion equation to simulate water and pesticide fluxes in the soil profile as well as the mobile/immobile phase to account for preferential flow. This is coupled to sorption and decay of chemicals depending on temperature and soil moisture.

7.5 KNOWLEDGE GAPS

The following knowledge gaps were identified through literature review:

- Pesticide runoff is generally modelled using semi-empirical methods and there is need to introduce a more mechanistic approach based on rainfall intensity.
- Volatilisation from plant and soil is only theoretically understood, mainly because of the difficulty in getting reliable measurements due to the very thin layer from which it occurs.
- The key parameter in the adsorption/desorption process is the sorption coefficient. This is specific for the pesticide species and needs to be determined experimentally.
- In order to model plant uptake of pesticides, an accurate plant water uptake subroutine is essential and this is often neglected in pesticide leaching models.
- The convection and convection-dispersion approaches require more validation work as these directly affect the accuracy of predictions of pesticide leaching. In some instances, detailed laboratory measurements need to be extrapolated to field conditions.
- Preferential flow is not fully understood and validated.
- Detailed, dedicated and scientifically sound models for predicting pesticide leaching are one-

dimensional and they assume there is no horizontal influx of water. For the purpose of this research, there is need to adapt existing models for the extrapolation to catchment scale. Knowledge gaps from microscopic to catchment level are related to convection, dispersion and lateral drainage flow, as well as adsorption and decay especially in the intermediate vadose zone.

- Chemical and microbial degradation (persistence) is a complex process and it is not an inherent property of the pesticide. The effect of factors controlling the degradation rate of pesticides is not always well understood (soil properties, weather conditions, properties of pesticide, etc.). In addition, theoretically sound relationships found in the laboratory may not always be applicable to field conditions.
- Similarly to volatilisation, photo-decomposition at plant and soil surfaces is only theoretically understood because of the difficulty in getting reliable measurements due to the very thin layer where it occurs.

7.6 RECOMMENDATIONS FOR FURTHER RESEARCH

Most validation exercises found in the literature are related to short-term experiments (e.g. one season) (Diekkruiger *et al.*, 1995). As the purpose of this research is to assess the impact of management practices on groundwater and catchment, it is important that model validation be done for a number of years. Alternatively, models validated over the short term could be used to run scenario simulations. For the purpose of scenario simulations, mechanistic models and model procedures should be preferred to empirical ones.

Spatial variability and preferential flow are issues that should be taken into account under certain management and environmental conditions, particularly the soil type (Bergstrom and Jarvis, 1994). Vanclooster *et al.* (2000a) recommended to run scenario simulations for worst, average and best cases, in order to overcome uncertainties.

Lack of good quality data sets is a common problem in modelling exercises. There is need to ensure that reliable input data sets are obtained. The number of input data sets and easiness of measuring the relevant input needs therefore to be balanced with quality.

It is also important that the model user is fully acquainted with the model to ensure quality control in the verification (code testing, debugging, etc.) and validation exercises.

7.6.1 Links to other non-point source contaminants from agriculture

Pesticide runoff should be modelled in conjunction with sediment transport, as portion of the transported pesticides is adsorbed onto solid particles of minerals and/or organic matter.

The process of volatilisation can be associated with losses of pesticide during spraying. As volatilisation mainly occurs from the soil surface where pesticides accumulate, this process can also be linked to sediment transport.

The adsorption/desorption of pesticides on solid particles can be associated with adsorption/desorption of nutrients (nitrogen, phosphorus) and heavy metals. The pesticide plant uptake, modelled as a function of plant water uptake, can also be associated with the nutrient (nitrogen, phosphorus) and heavy metal plant uptake. Similarly, pesticide fluxes in the soil profile, including preferential flow, as well as fluxes and processes occurring in the intermediate vadose zone, can be associated with nutrients (nitrogen, phosphorus) and heavy metals. Interactions between different pollutants should also be investigated.

On-farm water and pesticide management should be the cornerstone on which regional management is based. It is therefore essential to link processes occurring in the soil profile and intermediate vadose zone to groundwater models in order to estimate the impact of management practices on the catchment. The approach to be used will depend on the purpose and objectives of this research.

8. PUTATIVE MITIGATION MEASURES TO DECREASE POTENTIAL NON-POINT SOURCE POLLUTION FROM PESTICIDES IN AGRO-ECOSYSTEMS

8.1 PESTICIDE APPLICATION

8.1.1 Application technology

Pesticide Efficacy in Perspective

The chemical control of agricultural pests is one of the most important aspects of successful crop production in South Africa. Annually several millions of Rands are spent on chemical control of weeds, plant diseases and insect pests of agricultural crops. In practice fluctuations in efficacy occur concerning the degree of success of pest control, thus not only harming the trust in chemical control, but also in the present economic climate, the profitable production of agricultural crops.

The prime factor leading to poor chemical control lies in the application techniques used. Spray equipment and the application methods to control specific pests on crops do not always keep up with the practical recommended requirements, and leading to unsatisfactory results. Information locally obtained from the agrochemical industry shows that in South Africa more than 80 % of all the cases where poor control is experienced resulted directly to the following factors:

- a) Ineffective application techniques
- b) Condition of spray apparatus
- c) Unfavourable climatic conditions

Surveys done in the USA to evaluate pesticide application efficacy, showed that only one out of every three operators commercially applying pesticides stay inside the 5 % accuracy range of the recommended application requirements. In Germany out of a total of 510 sprayers in use by producers, only 40 % measured up to the application requirements. In the above-mentioned cases the major cause for poor application can be attributed to faulty apparatus or incorrect nozzles used during spraying.

Similar trends were found in South Africa following a survey undergone 1988 amongst South African producers.

Critical evaluations of all the factors leading to ineffective control subsequently identified according to importance are:

- Extension services to agricultural producers by either the private or agrochemical sectors.
- Training of pesticide application operators.
- Low standard of maintenance and calibration of application apparatus.

The overall impression indicates that effective pesticide application is a world-wide problem. Researchers are investigating the problem, not only with respect to cost efficient application, but also with respect to the safe use and protection of natural resources.

Chemical pest control

The important role that the agro-chemical industry plays as crop protector in agriculture cannot be emphasized enough. Without the help of chemical crop protection 65 % of the current total gross income of agricultural crops can be destroyed by pests. A disturbing factor is that despite the fact that

nearly R1 007 million (Table 8.1) are spend annually on pest control, a loss of more than R1 400 million is still lost due to ineffective control practices.

The estimated loss (Table 8.1) is mainly attributed to ineffective control of pests as a result of poor application using various types of spraying apparatus commonly used in South Africa (Table 8.2).

Table 8.1 Agrochemical costs to control agricultural pests in South Africa (Agricultural and Veterinary Chemical Association of South Africa 1993)

Weed control	R 338 million
Insect and mite control	R 223 million
Plant disease control	R 180 million
Adjuvants and related products	R 75 million
Application cost *	R 191 million
Total	R1 007 million

* Estimated value calculated at 25% of the pest control cost

Table 8.2 Projected yield loss attributed to ineffective pesticide application techniques

Crop	Loss	
	%	R (million)
Maize	25	434
Wheat	25	204
Grain sorghum	20	19
Peanuts	30	15
Sunflower	30	56
Soya beans	25	4
Cotton	30	38
Tobacco	20	35
Deciduous fruit	20	85
Vineyard	20	86
Citrus	25	67
Subtropical fruit	25	37
Potatoes	35	69
Tomatoes	30	32
Average	22	1 404

* Represent all agricultural crops cultivated in South Africa

Table 8.3 Percentage efficacy achieved with current type of spraying apparatus used in South Africa

Crop	Aerial	Boom and Nozzle	Mist blower
Maize	60-80	70-90	
Wheat	60-90	80-90	10-80
Gain sorghum	80-90	70-90	10-70
Peanuts	50-80	70-90	
Sunflower	60-80	60-80	10-80
Soya beans	60-90	70-90	60-90
Cotton	60-80	70-90	60-80
Tobacco	60-80	60-80	10-70
Deciduous fruit	0-20	40-90	80-90
Vineyard	0-60	70-90	70-90
Citrus	50-80	80-90	70-90
Subtropical fruit		10-90	60-80
Potatoes	60-80	75-90	10-80
Tomatoes			70-90
Average	60-80	70-90	50-90

* Represent all agricultural crops cultivated in South Africa

These statistics demonstrate the current situation in South Africa and stress the need for training and education of farmers and pest control operators to increase the standard of effective and safe application. Basic research and information concerning effective application must be done by research institutions such as Universities and Departmental Research institutes. The accurate transfer of this knowledge and high level of training is primarily the task of Education Organisations such as ascribed in courses for under graduate and post graduate studies by Universities, Agricultural colleges, Technikons and Technical colleges.

The finishing touches and implementation at farm level must take place under leadership and supervision of competent agricultural experts associated with state and private information services.

The key to successful implementation of environmentally friendly pesticide application practices is however in the hands of the producer. The escalation of production costs and the destruction of our natural resources, forces the agricultural industry to deal with this aspect as part of the production chain on a cost effective basis.

Management Skill

The key to successful pest control is emphasized in the standard of knowledge and education of the person responsible for decision making and the implementation of an application operation.

As manager, the producer or person responsible for the implementation of pest control must have a sound knowledge of the use and application of chemical pesticides. Decision making includes:

- a) The use and application of pesticides must be cost effective
- b) Irresponsible use of pesticides resulting in the build-up of pesticide resistance must be excluded
- c) Harmful and undesired side effects caused by the misuse of pesticides must be minimised

Pesticide costs

In most cases where chemical pest control is applied, the cost of the pesticide and application are relatively small in comparison with other production costs.

There are, however agricultural commodities, like deciduous fruit, viticulture and cotton to name only a few where pesticide control expenditures tend to be the most important production costs.

The increasing costs associated with research and development of new chemical pesticides do have a significant effect on the final production cost of a crop.

Application costs also increase as a result of higher fuel prices, and the development of more sophisticated spraying apparatus. Higher application costs are not normally correlated with higher crop yield, which emphasizes the need of training for cost effective pesticide application.

Pesticide resistance

Pests, like all other biological organisms are constantly adapting to changes in their environment. These evolutionary changes under "normal" pest control strategies takes place slowly as a result of pesticide selection pressure applied over many generations to the same pest complex.

The prolonged use of pesticides on a population may affect the population in two ways namely:

- a) Application of sub-lethal dosages of a pesticide shortens the period of time in which these adaptive changes take place, resulting in a total resistance build up of the population.
- b) Application of excessive amounts of pesticides has the consequence that the individuals in the population that by chance have the ability to survive excess applications are selected for subsequent generations therefore all have this ability to survive excess applications, and the population becomes in a relative short period uncontrollable.

To combat ineffective control, human reaction is normally either to increase the dosage rate, or to shorten the application periods between follow-up applications. Another alternative is to use pesticides or to mix pesticides of different chemical structures which resistance have not yet been built up. Such measures only provide a temporary solution and as a result of inadequate knowledge, these methods may lead to more serious resistant situations.

The consequence of pesticide abuse is clearly demonstrated in that more than 400 insect species in the world already built up resistance towards one or more chemical pesticide. Particularly dangerous is the situation where cross-resistance against a group with related chemical structures is formed; for example, against the various organophosphate compounds, or even worse, where super-cross-over resistance develops against pesticides of different chemical origins and toxicological effects; for example between chlorinated hydrocarbons and synthetic pyrethroids.

Once resistance has occurred, it is a very slow process to reverse resistance in a population back to the original level of susceptibility. It is therefore of utmost importance to develop and implement integrated pest control strategies that prevent the build-up of pesticide resistance.

Side-effects

Apart from the development of pesticide resistance, other important side-effects are:

a) Adverse effect on beneficial and non-target organisms.

Pest insects and a lot of other organisms exist according to a certain harmonical balance in the agro-ecosystem. Some of these organisms practice a natural influence on the control of pest populations or on other individuals by means of parasitism and/or predation. By the injudicious use of pesticides mainly the ones that have a long toxilogical after effect or those known as wide spectrum pesticides, the natural harmonical balance is disturbed by killing mainly the sensitive beneficial insects which contribute to the environmental equilibrium. Under these circumstances pest insects as well as other potential pests can reproduce undisturbed and developed to pest population levels. Side effects is not only limited to the sprayed target area. The drifting of droplets under especially unfavourable climatological conditions to non-target areas attribute to environmentally disturbance and especially to sensitive crops by herbicidal applications.

b) Human pollution

Most chemical pesticides which are toxic towards pests also holds in danger for humans, either by the use of sprayed products before the expiring date of the waiting period or by contamination of the body during application. Obliging safety measures must always be carried out by strictly following the instructions on pesticide labels.

8.1.2 Integrated Pest Management

The key solution to pesticide control lies in the strategy followed to control pest populations. The influence of the control strategy must have the least detrimental effect towards humans and the environment.

The principle of integrated pest management is well-founded on the fact that a chemical pesticide can only be used when it is absolutely necessary to avoid economically losses. It must justify costs involved and the substance used must be selective towards the pest and be environmentally friendly.

In the development of an integrated pest management program the following steps are necessary

- The knowledge of the biology and ecology of the pest in association with the biology of the crop which must be protected.
- Monitoring of pest populations
- Identification of target area or pest to be sprayed

Pest biology and ecology

For the control of any pest the correct specie identification is necessary. Near related pest species can seem identical and can reveal the same behaviour but can differ in their susceptibility and reaction towards specific pesticides. Knowledge of the pest's biology and behaviour present the opportunity to identify weak links in the life cycle of the pest for development of a optimum control strategy.

Knowledge of the population dynamics that is, the normal expected changes in the population density in reaction to key control factors effecting pest behaviour like the host plant as a source of nutrition, climatological conditions, natural enemies or other antagonistic organisms, all attributes to the correct selection and choice of a pesticide with respect to the mode of action nl. contact, oral and systematic to be used for effective control.

The biological and morphological development of the host crop on which the pest occurs is determinant with respect to the identification of the token area of the plant which must be protected. That requires knowledge of the vegetative and reproductive growth pattern of the plant as well as the growth tempo against which new unprotected growth developed after treatment.

Monitoring of pest populations

The monitoring of an insect population assures that precautionary measures for control actions could be made. That insures that pesticides are only applied when it is really necessary to prevent economical yield losses, instead of spraying according to a fixed spray program which may not coincide with a population built-up.

Pest populations are monitored by means of direct or indirect methods.

Direct method

Producers or pesticide application operators responsible for the monitoring of a pest population must be trained by means of specific methods to scout for a pest or a weed which must be controlled. The method of scouting must be easy and reliable and must preferably be done by a person not directly involved with the supplying or selling of the pesticide. The keeping of spray records before and after the application is of utmost importance to evaluate if the operation was successful. The keeping of a data bank of history of pest occurrence and control efficacy for a certain crop or pest in a certain area is vital for the development of a control strategy.

Indirect method

The indirect method of population monitoring makes use of key factors which effect changes in pest population occurrence and attack. These factors include for example changes in average day and night temperatures, rainfall patterns, presence of other host plants in or around the crop area, and the physiological attractiveness of the crop plant for the pest, e.g. when the plant is in the flowering stage.

Identification of target area

Before taking of a decision of any control strategy the target area to be protected or sprayed must be clearly identified to ensure that the pesticide is applied on the correct site and target area. Knowledge of the biology and behaviour pattern of the pest and its host crop is of importance especially with respect to the type of pesticide and spray apparatus which is to be used. These choices rest on the knowledge of all the factors which could have an influence on successful control.

8.1.3 Influence of Climatological Factors on Effective Pesticide Application

The only certainty of the weather is the uncertainty thereof. Nevertheless it fulfills an important role in the efficiency of pesticide application. The constant projection of climatological conditions is of the greatest importance for continued success. The use of pesticides, especially where herbicides are used on specific identified areas does cause tremendous problems for farmers, when drift takes place to non-target areas. Drift results in:

- Decreased efficacy
- Precipitation of chemicals in non-target areas which contributes to pollution and contamination of the environment.

Notwithstanding the method of application used, (aerial or ground equipment), drift from the target area will always take place.

Drift can be defined as follows:

Primary drift

Primary drift is the movement of droplets (water as carrier plus active ingredient of the pesticide) to a non-target area. Drift is the result of factors associated with the method of application. Important application factors like small droplets, spray release height and aspects like incorrect spraying pressure and spraying volumes applied, improves primary drift. For example, where ground

applicators are used, depending on cross-windspeed, droplets larger than 200 micron move less than 10 metre off target, while droplets smaller than 100 micron can move up to several thousands of metres under unfavourable conditions.

Factors responsible for primary drift can be identified and their effect on effective application can be managed simply by following the recommended climatological and application recommendations stated on the pesticide label.

Secondary drift

Secondary drift is related to the volatility of the pesticide chemical properties and is independent of the method of application. Secondary drift, sometimes known as vapour drift, cannot be controlled by the applicator. Vapour drift is difficult to explain, but may be due to evaporation of air-carried droplets or evaporation of pesticide precipitant from the target positions. Vapour drift is dependant on the vapour pressure of the chemical substance of the pesticide, but is also influenced by the soil, plant and air temperatures in and around the target area.

When a pesticide is applied surface area of a spray cloud of droplets increases dramatically if it is broken up into smaller droplets - especially if the droplets, are smaller than 50 micron. While in flight a droplet will lose its volatile substance due to evaporation over its surface area. The tempo of evaporation decreases, as the evaporated substance saturates the surrounding air. To control secondary drift, adjustments in the concentration of chemicals used in spraying formulations are important to suppress the vapour pressure of the spraying solution.

The disadvantage of water, the most important diluting agent used for pesticide spray mixtures, is that it is volatile and evaporates relatively fast. The life-time of a water droplet can be calculated as follows:

$$t = d^2 / 80 \Delta T$$

where t = life-time in seconds,
 d = droplet diameter (micron)
and ΔT = the difference in temperature ($^{\circ}\text{C}$) between a wet and dry bulb thermometer.
e.g. with $d = 100$ micron and $\Delta T = 5^{\circ}\text{C}$
 $t = (100)^2 / 80 \times 5 = 25$ seconds

Wind speed and direction

A droplet which is released in still air will accelerate downwards because of gravitational force. When gravitational force is balanced by aerodynamic forces, the droplet falls to the ground at a constant velocity which is defined as terminal velocity.

The terminal velocity of a 500 micron droplet is usually reached 70 cm after release, while the terminal velocity of a 100 micron droplet is reached within 2.5 cm from the point of release. Smaller droplets are thus more exposed to the influence of the wind for a longer period.

Although the loss of droplets because of drift is usually associated with higher wind speeds, drift can also take place at low wind speeds, especially where inversion conditions occur. The theoretical sideways movement of a 240 micron droplet which falls from a height of 3 m at different wind speeds under inversion conditions, is illustrated.

Theoretical distance m (metre) a droplet drift when released from a height of 3 m above the target area at different wind speeds

Wind speed (km/h)	1	2	4	8	16	32	64
Distance (m)	1.6	3.3	6.7	13.7	26.7	53.3	107

Turbulence which is usually associated with higher wind speeds, is the main cause of droplet drift. Turbulent air movement will affect some of the droplets more than other, as indicated in the above tables. Although it is difficult to quantify, it is generally expected that turbulent air movement pulls 50% of the applied spray cloud to the ground, while the remaining 50% drifts further away.

During field conditions, wind speed is never constant and an increase in wind speed is usually experienced with an increase in height above ground level.

Characteristic wind speeds according to the Beaufort scale and provides Pesticide Operators practical ways of determining wind speeds.

Wind strength	Description	Specification on land	Equivalent velocity at 10 m above the ground (average)		
			Knots	m/sec	km/h
0	Calm	Smoke rises vertically	1	0.2	1
1	Light air movement	Direction which smoke moves	1-3	0.8	1-5
2	Soft breeze	Wind felt on face, leaves move	5	2.4	6-11
3	Light breeze	Leaves move constantly	9	4.3	13-19
4	Moderate breeze	Leaves and dust blown up	13	6.7	20-28

Note: Meteorological wind readings are mostly done at a height of 10 m. Wind speed at 0.5-1.0 m above ground level is approximately half of that at 10 m.

The assistance of wind to improve penetration of droplets in the plant canopy will benefit deposition. It is difficult to specify the maximum wind speed for safe application conditions, as droplet size, height of spray release and spray pressure all have an effect on droplet drift. Good logic, self discretion and practical experience are the best guidelines to be followed when to apply a spray or to stop.

Air movement

The amount of pesticide spray that reaches the target area is greatly affected by factors such as the temperature, wind velocity, and wind direction. Information on the climate close to the ground is necessary to understand the effect of meteorological conditions on the movements of droplets.

Air temperature is influenced by atmospheric pressure, therefore air temperature decreases with height above ground level. This means that if a mass of air rises, without heat being added or removed, it will expand and cool. A decrease of temperature of 1°C for each 100 m above ground is known as the adiabatic lapse rate. If the decrease in temperature is more rapidly than 1°C for each 100 m a super-adiabatic lapse rate exists.

During the day the ground is warmed by the sun and it radiates heat energy to the air which is close to the ground. This air will start to rise and continue to do so while it remains hotter and lighter than the air around it. To replace the pocket left by the warm air, cool air will sink down between the columns of rising hot air. This convective movement of air creates an unstable atmosphere and turbulent conditions.

From late afternoon and during the night the ground is no longer warmed by the sun. As the ground loses heat by radiation it begins to cool and so will the air directly above the ground. Air temperature increases with height and an inversion condition exists. Inversion conditions occur in the evening if there is a clear sky after a hot sunny day and persist until the next morning when the sun heats up the ground again.

Fog or early morning mist occurs under inversion conditions of low wind velocity and smooth or laminar airflow. Fluctuating gusts and changes in wind direction may occur due to irregularities in the ground surface and this may destroy an inversion layer.

Thus atmospheric stability is influenced by the convection movement of air masses caused by thermal gradients, and surface friction of the local topography. We can thus distinguish three conditions of air movement by looking at the air temperature gradient, namely:

- Neutral stable conditions- no difference in temperature gradient
- Stable inversion conditions- increase in temperature with increase in height
- Unstable, adiabatic conditions- decrease of temperature with increase in height.

By measuring air temperature and wind velocity at specific heights one can calculate a stability ratio (SR) which is as follows:

$$SR = \frac{T_1 - T_2}{U^2} \times 100\,000$$

Where: T_2 = temperature ($^{\circ}\text{C}$) at 10 m above ground level

T_1 = temperature ($^{\circ}\text{C}$) at 2.5 m above ground level

U = wind velocity (cm/sec) at 5 m above ground level

A positive stability ratio indicates stable temperature inversion conditions and a negative ratio, unstable adiabatic conditions.

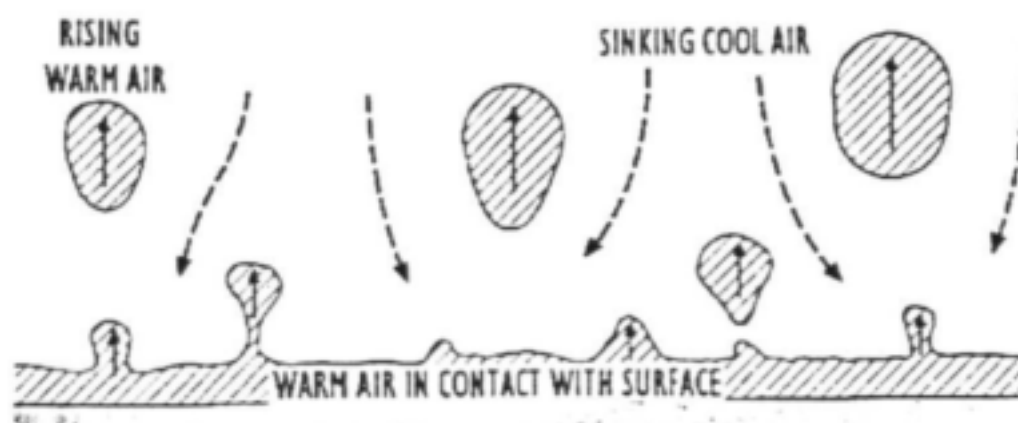


Figure 8-1 Dynamics of warm and cold air movement above the earth's surface

Relative humidity

Relative humidity (RH) influences the tempo of evaporation of water which generally is used as carrier of the droplets. The lower the RH, the higher the evaporation and the faster the droplet becomes

smaller, falling slower to the ground and under windy conditions drifts further away. The table below illustrates the time-lapse and distance that pure water droplets fall until they have lost 90% of their volume because of evaporation under different RH- conditions at 26°C.

Initial droplet size (μm)	Final droplet size (μm)	Relative humidity			
		30%		70%	
		Time (sec)	Distance (m)	Time (sec)	Distance (m)
100	46	4.2	.75	9.2	1.60
80	37	2.8	.25	6.3	.67
60	28	1.7	.13	3.8	.23
40	19	0.8	.10	1.8	.15

Droplets larger than 150 - 200 micron are not affected to such a great extent by a low RH, because they reach their target area before a meaningful loss of volume has taken place.

Droplet distribution

Directly after droplets have left a spray nozzle, they begin to lose momentum and move slower to the target area. The effect of wind on small droplets in the spray cloud can influence the initial number of droplets directed towards the target area by drifting them away. The higher the spray boom or the aircraft is above the target area, the greater the possibility that droplets will be carried away by the wind and may never reach the crop to be treated. The drift and precipitation of small droplets down wind is proportional to the spray nozzle height above the target and the strength of the wind speed.

This droplet behaviour and characteristic can be applied to one's advantage, where a ULV hand-application apparatus is used. In this case, a light wind can be depended upon to distribute droplets by drifting it into and over the target area. With the use of a non-volatile formulation, e.g. oils, the evaporation of droplets is limited and the drift of the droplets can be applied to great advantage.

Climatological guidelines for spraying

For effective pesticide application give special attention to the following conditions:

- Stop spraying if there are sudden blasts of wind, or if the wind speed is higher than 15 km/h for insect and plant disease control. Where herbicides are applied, wind speeds higher than 8 km/h must be avoided.
- Application must be stopped when the wind direction changes in the direction of a susceptible crop.
- Evaporation plays a large role in droplet size. Stop spraying if there is a difference of 8°C between the wet and dry bulb thermometer readings. If an anti-evaporation adjuvant has been added to the spraying mixture, a difference of 12 °C can be allowed. Producers are warned that this measure alone will not ensure effective coverage. Other factors like high soil temperature can also influence droplet distribution through the formation of convection currents.
- Plants which are wet with dew, rain or irrigation water must not be sprayed.
- When using systemic pesticides, it is essential that the bottom parts of the plants are thoroughly wetted with the spray solution. The action of systemic pesticides is to move in most cases upwards in plants. Do not spray wilted plants.

8.2 SURFACE RUN-OFF

Recent research has increasingly focussed on identifying mitigation measures capable of reducing the risk of exposure of surface waters to pesticide contamination. In general different risk mitigation options and the understanding of the impact of mitigation measures on pesticide exposure are poorly developed and it is clear that further research in this area is required (FOCUS, 2001). The implementation of best management practices (BMP) are management driven decisions or precautions that are implemented by the farmer, in an attempt to reduce chemical transport from croplands into non-target areas. With respect to reducing pesticide loss via runoff, conservation tillage has been shown to reduce pesticide loss in runoff, with contour tillage resulting in a 10% reduction compared to up-down-slope tillage (Felsot *et al.*, 1990). Other more simple strategies involve planning the application of pesticides in relation to forecasting of significant rainfall events. Application just before a heavy rainfall event will result in large losses of pesticides and may reduce the efficacy of the pesticide as a result of it having been washed off the target crop. This may also result in the pesticide having to be re-applied, leading to increased economic costs and risk to the environment.

Vegetated buffer strips bordering agricultural plots have been shown to be effective at reducing the level of pesticides in surface runoff (Patty *et al.*, 1997) and help improve the water quality and remove sediment from surface runoff by stabilizing stream-banks and promoting infiltration (Castelle *et al.*, 1994). The vegetation helps to resist the formation of channels (erosion rills), resulting in low flow rate sheet-flow as opposed to rapid channelised flow, which allows more time for the settling of sediments, infiltration and adsorption of pesticides to vegetation.

In more developed countries there is a movement towards investigating mechanisms by which different agricultural water body types (i.e. ditches, streams and ponds) can mitigate pesticide exposure, so as to characterise their vulnerability towards different pesticide inputs (i.e. runoff, spray drift or drainage). Two important characteristics that have a significant impact in reducing the presence and transport of pesticide concentrations are the hydrology of the water body and the presence or absence of aquatic macrophytes in these systems, as both have a profound effect on the extent and duration of exposure following a non-point source event. In this respect, wetlands and slow flowing agricultural ditches have been the subject of increased research attention. This has led to the realisation of the importance that aquatic macrophytes may have in mitigating risk through the development of substrates and habitat and aiding in increasing potential for dissipation. In recent years, increased attention has focused on the ability of aquatic macrophytes to reduce the transport of pesticides through constructed wetlands and in agricultural ditches (Hand *et al.*, 2001; Moore *et al.*, 2001a).

Studies have shown that vegetated wetlands can reduce pesticide concentrations as well as toxicity towards macro-invertebrates. Two intensive studies have focused on the mitigation potential of a constructed flow-through wetland that was built into a tributary that flows into the Lourens River (Schulz *et al.*, 2003; Schulz and Peall, 2001). The wetland has an inlet that receives water from a tributary that flows directly through an intensive fruit orchard area and an outlet, which flows directly into the Lourens River mainstream. Field based monitoring showed a decrease in concentrations of runoff-related pesticide contamination (dissolved and particle-associated), nutrients and suspended sediment as well as a decrease in toxicity from the inlet to the outlet during a large rainfall event (Schulz and Peall, 2001). Reduced flow conditions in the wetland promote siltation, resulting in decreased turbidity and thus a decrease in sediment associated pesticide concentrations. Furthermore, the reduced flow slows the transport to such an extent that there is more time to allow for degradation of pesticides via volatilization, photolysis, hydrolysis or metabolic degradation (Schulz *et al.*, 2003). Vegetation within the wetland also provides additional substrate for the adsorption of aqueously dissolved pesticides after which they are rapidly broken down (Schulz *et al.*, 2003; Moore *et al.*

et al., 2001a). In general, constructed wetlands have been shown to be very effective in mitigating runoff related pesticide contamination and have been suggested as best management options in agricultural areas to ensure protection of sensitive or more vulnerable water bodies. Similarly, ditches adjacent to agricultural fields have also been shown to minimise pesticide transport via adsorption to macrophytes (Moore *et al.*, 2001b). However, it is important to note that their effectiveness in reducing pesticide transport may be dependent on the cause of contamination (i.e. spray drift or runoff). A study conducted in a slow flowing stream that flows into the Lourens River has shown that vegetation within the ditch is very effective in reducing spray drift events and runoff events resulting from 10mm rainfall. However the ditch was ineffective during a 22mm rainfall event, as a result of significantly increased discharge (results being prepared for publication).

8.3 MITIGATING LEACHING AND PERSISTENCE

8.3.1 Bioremediation and Phytoremediation Technology

Bioremediation as defined by the American Academy of Microbiology is "the use of living organisms to reduce or eliminate environmental hazards resulting from accumulations of toxic chemicals or other hazardous wastes" (Gibson and Saylor, 1992). Bioremediation is not a new technology. This is evident in that humankind has practiced composting, sewage treatment and fermentation since the beginning of recorded history. All of these processes utilize microbial processes in a degradation process. The modern use of bioremediation began with the opening of the first biological sewage treatment plant in Sussex, UK, in 1891. The use of this technology in cleaning up pollutant spills is gaining popularity. Over the past ten years an increase in the types of contaminants to which bioremediation is applied has been evident.

Indigenous and enhanced micro-organisms have been shown to degrade industrial solvents, polychlorinated biphenyls, explosives and many different agricultural chemicals, including pesticides. Pilot, demonstration, and full-scale applications of bioremediation have been carried out, but on a limited scale. Internationally micro-organisms have been identified and employed, to transform and degrade contaminants. Admittedly the application of this technology to *in situ* bioremediation of polluted sites has been limited. At present, bioremediation is often the preferred method for remediation of especially petroleum hydrocarbons, because it is cost effective, and it converts the petroleum hydrocarbons into the harmless by-products carbon dioxide and water.

Bioremediation can be used to degrade concentrated organic contaminants near their source or as a secondary remediation strategy following specific physical or chemical methods. It can also be applied for sequestration of metals and radio-nuclides through microbially mediated transformation processes as well as for remediation of large plumes of diluted contaminants that are broadly dispersed in the environment.

However, the full benefits of bioremediation have not been realized because processes and organisms that are effective in controlled laboratory tests are not always equally effective in full-scale applications. The failure to perform optimally in the field setting stems from a lack of predictability due, in part, to inadequacies in the fundamental scientific understanding of how and why these bioremediation processes work (Forsyth *et al.*, 1995).

Micro-organisms degrade or transform contaminants by a variety of mechanisms. Petroleum hydrocarbons for example are converted to carbon dioxide and water or are used as a primary food source by bacteria, which use the energy to generate new cells. Where the hydrocarbons are chlorinated the degradation takes place as a secondary or co-metabolic process rather than a primary metabolic process. In such a case enzymes, which are produced during aerobic utilization of carbon sources such as methane, degrade the chlorinated compounds. Under aerobic conditions, a

chlorinated solvent such as trichloroethylene can be degraded through a sequence of metabolic steps, where some of the intermediary by-products may be more hazardous than the parent compound (e.g. vinyl chloride).

Certain pollutants like metals and radio-nuclides cannot be degraded through biological activity, but can be transformed from one chemical to another, or move out of the soil by plants to above and below ground plant tissue. Some fungi, can convert dissolved arsenic and selenium to gaseous forms through methylation, and in this way remove the compound from soil. Bacteria have been shown to change the oxidation state of heavy metals such as chromium, selenium and mercury as well as radio-nuclides (uranium) by using these as electron donors or acceptors in the metabolism.

The impressive capabilities of micro-organisms and plants to degrade and transform contaminants should provide tremendous benefits in the clean-up of pollutants from spills and storage sites. These remediation ideas have provided the foundation for many *ex situ* waste treatment processes (including sewage treatment) and a host of *in situ* bioremediation methods that are in practice today (Hinchee *et al.*, 1994).

Explosives and polychlorinated biphenyls have been biodegraded in field scale experiments (Harkness *et al.*, 1993). Bacterial transformation and immobilization of trace metals such as selenium also has been implemented on limited basis (Benson *et al.*, 1993).

Over the past ten years, progress has been made in expanding the number and type of contaminants to which bioremediation can be applied and in the number of practical methods for implementing *in situ* bioremediation. Techniques such as hydrofracturing have been developed for improved delivery of nutrients to micro-organisms in low-permeability geologic media. Methods have been developed for creating passive treatment systems such as bio-filters (Taylor *et al.*, 1993). Novel concepts for using microbial produced biopolymers as *in situ* plugging agents have also been explored (Li *et al.*, 1994). Bioremediation is a more cost effective method of remediation as compared to incineration or physical and chemical remediation methods (Saaty and Booth, 1994; Wijesinghe *et al.*, 1992; Atlas, 1995). This technology has the potential to be one of the most cost effective technologies for dealing with environmental remediation problems.

8.3.2 Microbial Biodegradation – mechanisms

Biodegradation involves chemical transformations mediated by micro-organisms that:

- satisfy nutritional requirements
- satisfy energy requirements
- detoxify the immediate environment
- or occur fortuitously such that the organism receives no nutritional or energy benefit (Stoner, 1994).

Mineralisation is the complete biodegradation of organic materials to inorganic products, and often occurs through the combined activities of microbial consortia rather than through a single micro-organism (Shelton and Tiedje, 1984).

Co-metabolism is the partial biodegradation of organic compounds that occurs fortuitously and that does not provide energy or cell biomass to the micro-organisms. Co-metabolism can result in partial transformation to an intermediate that can serve as a carbon and energy substrate for micro-organisms, as with some hydrocarbons, or can result in an intermediate that is toxic to the transforming microbial cell, as with trichloroethylene and methanotrophs.

The two main classes of biodegradation reactions are aerobic and anaerobic reactions

Aerobic biodegradation involves the use of molecular oxygen (O_2), where O_2 (the "terminal electron acceptor") receives electrons transferred from an organic contaminant:



Thus, the organic substrate is oxidised (addition of oxygen), and the O_2 is reduced (addition of electrons and hydrogen) to water (H_2O). In this case, the organic substrate serves as the source of energy (electrons) and the source of cell carbon used to build microbial cells (biomass). Some micro-organisms (chemo-autotrophic aerobes or litho-trophic aerobes) oxidize reduced inorganic compounds (NH_3 , Fe^{+2} , or H_2S) to gain energy and fix CO_2 to build cell carbon:



At some contaminated sites, as a result of consumption of oxygen by aerobic micro-organisms and slow recharge of O_2 , the environment becomes anaerobic (lacking O_2), and mineralisation, transformation, and co-metabolism depend upon microbial utilization of electron acceptors other than O_2 (anaerobic biodegradation). Nitrate (NO_3), iron (Fe^{+3}), manganese (Mn^{+4}), sulphate (SO_4), and carbon dioxide (CO_2) can act as electron acceptors if the organisms present have the appropriate enzymes (Sims, 1990). JP-4 jet fuel constituents have been observed biodegraded, but only in the presence of NO as the electron acceptor (Hutchins *et al.*, 1991). Iron and manganese could also act as important microbial electron acceptors at soil concentrations ranging from 20 to 3,000 mg/kg for Mn and 3.8 to 5.2 percent for iron. Biodegradation of pentachlorophenol (PCP) has been observed to increase the presence of added Mn (Petrie *et al.*, 1995).

8.3.3 Environmental factors affecting bioremediation

Microbial ecologists have identified ranges of critical environmental conditions that affect the activity of soil micro-organisms. The conditions are summarised in the table below.

Critical Environmental Factors for Soil Microbial Activity (US EPA, 1989)

Environmental factor	Optimum levels
Oxygen	Aerobic metabolism: more than 0.2 mg/l dissolved oxygen Anaerobic metabolism: less than 0.2 mg/l dissolved oxygen
Nutrients	Sufficient nitrogen, phosphorus and other nutrients
Moisture	Unsaturated soil: 25-85% of water holding capacity will affect oxygen transfer into soil (aerobic status) Saturated soil: water will affect transport rate of oxygen and affect rate of aerobic remediation
Environment pH	5.5 - 8.5
Environment temperature	15 - 45°C

Many of these environmental conditions can be controlled and managed to enhance the biodegradation of organic constituents.

8.3.4 Phytoremediation

Phytoremediation is the use of vegetation for *in situ* treatment of pollutant contaminated soils, sediments, and water. It is best applied at sites with relatively shallow contamination of pollutants that are amenable to Phytotransformation, Rhizosphere Bioremediation, Phytostabilization, Phytoextraction and/or Rhizofiltration. This technology is especially valuable where the contaminated soils are fragile, and prone to erosion. The establishment of a stable vegetation community stabilises the soil system and prevents erosion. This aspect is especially relevant to Southern African soils where removal of large volumes of soil destabilises the soil system, which leads to extensive erosion.

Phytotransformation and rhizosphere bioremediation are applicable to sites contaminated with organic pollutants that include pesticides. It is an emerging technology that should be considered for remediation of contaminated sites because of its cost effectiveness, aesthetic advantages, and long-term applicability (Brown, 1995).

Plants have shown the capacity to withstand relatively high concentrations of organic chemicals without toxic effects, and they can uptake and convert chemicals quickly to less toxic metabolites in some cases. In addition, they stimulate the degradation of organic chemicals in the rhizosphere by the release of root exudates, enzymes, and the build-up of organic carbon in the soil.

8.3.5 Limitations

There are limitations to the technology that need to be considered carefully before it is selected for site remediation. These include:

- limited regulatory acceptance
- long time periods may be required for clean-up to below acceptable thresholds
- difficulty in establishing and maintaining vegetation at some sites

Recent field tests of phytoremediation are reported on wastes containing petroleum hydrocarbons such as benzene, toluene, ethylbenzene, and xylenes (BTEX) and polycyclic aromatic hydrocarbons (PAHs), pentachlorophenol, polychlorinated biphenyls (PCBs), chlorinated aliphatics (trichloroethylene, tetrachloroethylene, and 1,1,2,2-tetrachloroethane), ammunition wastes (2,4,6-trinitrotoluene or TNT, and RDX), metals (lead, cadmium, zinc, arsenic, chromium, selenium), pesticide wastes and runoff (atrazine, cyanazine, alachlor), radionuclides (cesium-137, strontium-90, and uranium), and nutrient wastes (ammonia, phosphate, and nitrate).

Different species of plants have been used in various applications including: *Salix* spp. (hybrid poplars, cottonwoods, and willow), grasses (rye, Bermuda grass, sorghum, fescue, bullrush), legumes (clover, alfalfa, and cowpeas), aquatic plants (parrot feather, duckweed, arrowroot, cattail, pondweed), and hyperaccumulators for metals (sunflowers, Indian mustard, and *Thlaspi* spp.).

Phytoremediation has successfully been applied at a brownfields site for remediation of soil contaminated with lead; a small pond at Chernobyl with uranium contamination; a riparian zone buffer strip at Amana, Iowa for nitrate and atrazine removal from agricultural runoff; and at an engineered wetland at Milan, Tennessee for TNT removal. In addition, many successful applications have involved remediation actions at small sites, such as agricultural cooperatives with pesticide and ammonia spills (Perchovich et al, 1996).

8.3.6 Potential application

Potential applications include:

- phytotransformation of petrochemical sites and storage areas
- ammunition wastes
- fuel spills
- chlorinated solvents
- landfill leachates (including biochemical oxygen demand (BOD) and chemical oxygen demand (COD))
- agricultural chemicals (pesticides and fertilizers)

Very often, phytoremediation is not the sole treatment option, but rather it is used in conjunction with other approaches such as removal actions or *ex situ* treatment of highly contaminated wastes, or as a polishing treatment. Plants are able to take-up contaminants directly from the soil water or release exudates that help to degrade organic pollutants via co-metabolism in the rhizo-sphere. Direct uptake of organics by plants could be an efficient removal mechanism from sites contaminated at a shallow depth with moderately hydrophobic organic chemicals.

Once the contaminant has been taken up and translocated, the plant could integrate the chemical and its fragments into new plant structures via lignifications (covalent bonding of chemical or its fragments into lignin of the plant). It is also possible that the plant could volatilise, metabolise, or mineralise the chemical completely to carbon dioxide and water. Chlorinated aliphatic compounds such as trichloroethylene (TCE) have been reported to be mineralised to CO_2 and less toxic aerobic metabolites (trichloroethanol, trichloroacetic acid, and dichloroacetic acid by Newman *et al.* 1997).

Another form of phytotransformation is phytovolatilization, whereby volatile chemicals or their metabolic products are released to the atmosphere through plant transpiration. Many organic chemicals that are recalcitrant in the subsurface environment react rapidly in the atmosphere with hydroxyl radicals, an oxidant formed in the photochemical cycle. The transfer of contaminants from the soil or groundwater to the atmosphere is not as desirable as *in situ* degradation, but it may be preferable to prolonged exposure in the soil environment and the risk of ground-water contamination.

Nitroreductase and lactase enzymes in plants can break down ammunition wastes such as TNT (2,4,6-trinitrotoluene), and they may incorporate the broken ring structures into new plant material or organic detritus that becomes a part of sediment organic matter. Detoxification mechanisms may transform the parent chemical to non-phytotoxic metabolites that are stored in plant tissues (Schnoor *et al.* 1995). A thorough understanding of pathways and end-products of enzymatic processes will simplify toxicity investigations of *in situ* phytoremediation.

8.3.7 Rhizosphere Bioremediation - Interaction Between Plants and Micro-organisms

Phytoremediation in the rhizosphere increases soil organic carbon, soil bacteria, and mycorrhizal fungi, all factors that encourage degradation of organic chemicals in soil. Rhizosphere bioremediation is also known as phytostimulation or plant-assisted bioremediation. Jordahl *et al.* (1997), showed that the numbers of beneficial bacteria increased in the root zone of hybrid poplar trees relative to an unplanted reference site. Plants may also release exudates to the soil environment that help to stimulate the degradation of organic chemicals by inducing enzyme systems of existing bacterial populations, stimulating growth of new species that are able to degrade the wastes, and/or increasing soluble substrate concentrations for all micro organisms (Barkovskii *et al.* 1996).

Plants help with microbial transformations through:

- mycorrhizal fungi and bacteria associated with plant roots metabolise the organic pollutants
- plant exudates stimulate bacterial transformations (enzyme induction)
- build-up of organic carbon increases microbial mineralization rates (substrate enhancement)
- plants provide habitat for increased microbial populations and activity
- oxygen is pumped to roots ensuring aerobic transformations

Fungi, growing in symbiotic association with the plant, have unique enzymatic pathways that help to degrade organics that could not be transformed solely by bacteria. In addition to soluble exudates, the rapid decay of fine root biomass can become an important addition of organic carbon to soils that

serves to retard organic chemical transport. Microbial mineralisation of atrazine is directly related to the fraction of organic carbon in the soil (McFarlane *et al.*, 1987).

8.3.8 Natural and Accelerated Bioremediation

Intrinsic bioremediation can be defined as the combined effect of natural destructive and non-destructive processes to reduce a contaminant's mobility, mass and associated risk. Non-destructive mechanisms include sorption, dilution and volatilisation. Destructive processes are aerobic and anaerobic biodegradation.

Intrinsic aerobic biodegradation is well documented as a means of remediating soil and groundwater contaminated with fuel hydrocarbons. McAllister *et al.*, 1995; Brown *et al.*, 1995, Barker *et al.*, 1995, state that intrinsic aerobic degradation should be considered an integral part of the remediation process. There is growing evidence that natural processes influence other chemicals, such as aromatic hydrocarbons, chlorinated organics and mixed hydrocarbons. Studies by Ginn *et al.*, 1995 and King *et al.*, 1995, deal with intrinsic processes that immobilize and biodegrade aromatic hydrocarbons.

Bio-stimulation is the addition of nutrients, oxygen, or other electron donors and acceptors to increase the number or activity of naturally occurring micro-organisms available for bioremediation. Successful application of bio-stimulation was demonstrated at the DuPont Niagara Falls Plant manufacturing various organic and inorganic chemicals (Buchanan *et al.*, 1995). Chlorinated solvents were produced from 1930 to 1975. In-field evaluations determined that biological reductive anaerobic dechlorination was occurring naturally. A field program was implemented in a pre-selected area of the plant through use of *in situ* borehole bioreactor to attempt to stimulate indigenous biological reductive dechlorination by the addition of yeast extract (substrate) and sulphate (electron acceptor). A very active microbiological population developed, which reduced the *in situ* concentrations of chlorinated compounds by more than 94%. Concentration of the typical biological degradation products did not increase, probably due to alternative biodegradation pathways.

Bio-augmentation is the addition of micro-organisms that can bio-transform or biodegrade a particular contaminant. Fungi and bacteria present in contaminated soils or water cause bioremediation. Bacteria are usually the organisms of choice because:

- they have more rapid metabolic rates
- numerous metabolic pathways of various organic pollutants in bacteria have been determined
- bacteria can be genetically manipulated to improve their bioremediation capabilities (Bouwer and Zehner, 1993).

The use of bacteria also has many disadvantages. Most bacteria survive and metabolise best at pH 6.5 to 7.5 and many contaminated media have low pH. Recently, much research (Barr and Aust, 1994, Burton *et al.*, 1998) has been performed with *Phanerochaete chrysosporium* and other white-rot fungi because of the wide substrate range of the ligninolytic enzymes, which are capable of degrading chlorinated solvents, aromatic pollutants, pesticides, etc.

The major drawbacks of fungi are:

- the requirement for large amounts of co-substrate, such as simple sugars or cellulose,
- their relatively low growth rate
- the cost to produce fungal inocula is high compared to bacteria

Micro-organism strains can be improved by adaptation or mutagenesis to obtain more favourable characteristics.

Bio-augmentation products are made by large scale fermentation of single strains or simple mixed cultures (consortia) with desirable degrading properties. For dry products the fermentation mixture is dried on a suitable support, for liquid products organisms are suspended in a stabilizing solution. Commercial products are blended to provide an inoculum that will degrade a spectrum of contaminants likely to occur together. Products may be targeted to the contaminants usually produced by specific industries, to complex contaminants such as petroleum fuels, chlorinated solvents or pesticides. Composition of commercial products is usually closely guarded, but reputable companies should be able to produce degradation data that can demonstrate their product's applicability to a particular situation or project (Molnaa *et al*, 1989; Venosa *et al*, 1992).

8.3.9 In Situ versus Ex Situ Bio-remediation

In situ bio-remediation refers to below ground methods applied at the site of contamination whereas *ex situ* refers to above-ground bio-remediation, where the sediment or water has been extracted from the sub-surface.

The most widespread use of *ex situ* bio-remediation is the cleanup of storage tank leaks and oil spills in pipelines, tank farms and petroleum refineries. The technique, known as a solid-phase bio-remediation is similar to composting, in which soils are physically and chemically manipulated to stimulate breakdown of target organics by resident and exogenous microbes. In soil piles it is necessary to install piping networks to distribute air and allow addition of water and supplemental additives. Application of nutrients, pH regulating chemicals, and microbial strains may be performed by spraying (in a water slurry) or direct application with tillage to distribute the additives into the soil. For more heavily contaminated soils and sludges, slurry processes are used. These are modified versions of the activated sludge process used in various industrial wastewater treatment plants. The soil or sludge is fluidised with water and treated in an on site bioreactor. These systems are easily monitored and addition of nutrients and microbial strains can be made to closely control and enhance the bioprocess.

The third major category of on site bio-remediation involves pumping groundwater to the surface for treatment in an above-ground bioreactor. The effluent from the bioreactor, containing oxygen, nutrients and acclimated micro-organisms, is then injected back into the ground to remediate the contaminated soils associated with the groundwater. This process combines *ex situ* groundwater treatment with *in situ* soil treatment.

In situ bio-remediation has several advantages over *ex situ* techniques:

- In situ bio-remediation can be used to completely degrade and detoxify some organic contaminants, thereby permanently removing liability for the contaminants.
- For deep, widely dispersed plumes of heavy metals and radionuclides, *in situ* bio-remediation and immobilization may be the only viable solution.
- For some types of contaminants, physical and chemical methods of remediation may not completely remove the contaminants, leaving residual concentrations that are above regulatory guidelines. Bio-remediation can be used as a cost-effective secondary treatment scheme to decrease the concentration of contaminants to acceptable levels. In other cases, bio-remediation can be the primary treatment method, and followed by physical or chemical methods for final site closure.
- In some cases, natural attenuation (including natural bioremediation) of the contaminant plumes may be the only cost effective solution.

- For complex mixtures of contaminants requiring a combination or sequence of physical and chemical remediation methods, bio-remediation techniques that use microbial consortia to concurrently address all contaminants may be faster and more cost-effective.
- For very large sites.

8.3.10 Bio-degradation of Pesticides

Microbial degradation is an important step in the disappearance and, in most cases detoxification of pesticides. Herbicide biodegradation may prevent the problem of environmental pollution but it can also reduce the effectiveness of a compound in controlling targeted pests. Many soil applied pesticides are degraded more rapidly following repeated application at the same site (Kauffman 1987; Racke and Coats, 1990).

In a survey of soils from commercial fields, there was evidence that enhanced bio-degradation of the compound has been induced by normal field applications, in some soils by a single previous treatment (Walker *et al.*, 1993). Several herbicides are prone to degradation, including members of the thiocarbamate group, the ureas, linuron and monolinuron, the amides, propyzamide and napropamide, and the triazinones, chloridazon and metamiltron (Roeth, 1986; Walker and Welch, 1991; Roberts *et al.*, 1991).

Factors responsible for the enhanced degradation are micro-organisms present in the soil, able to degrade applied pesticide. Walker *et al.*, 1993, gave evidence, that bacteria are responsible for the process. The rate of degradation in soil was unaffected by treatment of the soils with the antifungal antibiotic cycloheximide, but was inhibited by the antibacterial antibiotic chloramphenicol. Usually mixed cultures of bacteria able to degrade pesticides are isolated from these soils by enrichment culture.

Early research indicated that napropamide was relatively stable to degradation in the soil, but the results of recent laboratory and field experiments have indicated that the rate of degradation is increased significantly in the soil treated previously with the same compound. Mixed cultures of bacteria were isolated from the soils by enrichment culture on a liquid mineral medium with napropamide as a sole carbon and nitrogen source (Walker *et al.*, 1993). The related herbicide diphenamid is also susceptible to enhanced degradation, suggesting that substituted amides may be particularly prone to this phenomenon.

Enhanced degradation of linuron occurred in soils, following repeated applications of the herbicide. These effects were more pronounced in laboratory incubations than in field persistence studies. Direct isolation of bacteria from soil failed to identify bacteria with the ability to degrade the herbicide. Sequential enrichment culture in a mineral base medium led to the isolation of an apparently stable mixed culture of bacteria, which would degrade the herbicide (Roberts *et al.*, 1991).

Metamiltron is a selective, pre and post-emergence herbicide, used for weed control in sugar beet, which acts by inhibiting electron transport in photosynthesis. The compound is degraded to desaminometamiltron in various soils and by different soil microbes. Parekh *et al.*, 1994, isolated metamiltron-degrading bacteria *Rhodococcus* sp. from the treated soils by enrichment culture. The herbicide was completely degraded within 7 days at 25° C in laboratory tests. A pure culture of *Arthrobacter* sp. was shown to degrade metamiltron, in the presence of alternative sources of carbon and nitrogen, within two weeks of incubation in the dark (Engelhardt *et al.*, 1982).

Bacteria capable of degrading carbofuran as sole carbon and nitrogen sources were isolated from liquid cultures of treated soils (Parekh *et al.*, 1994). Similar types of carbofuran-degrading bacteria were isolated from different soils. All isolates were Gram negative, aerobic rods that hydrolyzed the

insecticide to carbofuran phenol. Racke and Coats, 1990, imply the presence of carbofuran degrading bacteria in soils without a history of previous treatment. It is possible that many soils contain a small population of bacteria that have the ability to degrade certain compounds. The presence of such bacteria in untreated soils could be due to contamination from surrounding treated areas or they may be an inherent part of the soil microflora, which degrade naturally occurring analogous compounds. Chary *et al*, 1992, detected carbofuran degrading bacteria in 48% of samples from previously untreated plots.

Walker *et al*, 1996, isolated a mixed microbial culture able to degrade the herbicide to a single degradation product – naphthoxypropionic acid. Addition of this culture to a previously untreated soil introduced rapid degrading ability. These results gave hope for use of augmentation for the treatment of sites contaminated by carbofuran and other pesticides.

Atrazine is a broad-leaf, pre-emergence herbicide. It is a leading member of a class of triazine ring-containing herbicides that includes simazine and terbuthylazine. Atrazine has been found to be less biodegradable than other pesticides. However, a number of different bacteria have been identified that are capable of metabolizing atrazine to ammonia and carbon dioxide. Zeng *et al* (1999) presented atrazine degradation pathway map (Table 8.4). Organisms that can initiate the pathway are given, but other organisms may also carry out later steps. This pathway demonstrates microbial biodegradative diversity.

With some chemicals in some soils the waste is consumed completely, or has been reduced to an acceptable concentration level. In other kinds of soils, or when the toxic agents have low water solubility, or have been adsorbed in the soil matrix to a high degree, the micro-organisms would not be able to achieve sufficient degradation.

Successful bio-remediation of herbicides may be accomplished by installing plants at contaminated sites. Micro-organisms present or introduced to the rhizosphere degrade contaminants and plants provide nutrients, water and oxygen for micro-organisms. This method is explained in the chapter "Rhizosphere phytoremediation – interaction between plants and micro-organisms".

In soils contaminated with pesticides, a unique problem exists because of the presence of herbicides; chemicals designed to prohibit the growth of vegetation. Herbicide tolerant plants can survive at these sites and are ideal candidates for testing whether vegetation can be used to enhance microbial degradation of pesticide wastes.

Anderson *et al* (1994), reported on enhanced degradation of a mixture of three pesticides, namely the three herbicides atrazine, metachlor and trifluralin in the rhizosphere of a herbicide-tolerant plant - *Kochia* sp., growing on contaminated site. Enhanced degradation was observed in the rhizosphere soil after 14 days of incubation. Micro-organisms in non-vegetated soil also showed the ability to degrade the three compounds, but not to the extent of the rhizosphere soil.

expected at storage sites, can be expected to have leached into the soils significantly. Although leaching studies have not been conducted on specific POPs trials have been conducted for some pesticides used in the country. The herbicide tebuthiuron for example was found to leach significantly to a depth of 1.2 m within 18 days of its application and 30 mm rainfall (Meinhardt, 2003). Results of modelling leaching of this compound indicate that the compound could be leached to a depth of 3 m within a one year period. High soil mobility will also lead to horizontal spreading of a pollutant over a relatively large area, in comparison to the applied area, or extent of the spillage site. This kind of leaching patterns could also be expected from many of the POPs and obsolete pesticide stocks targeted for destruction.

In addition many South African soils are fragile soils which are prone to erosion should vegetated zones be disturbed or removed. If all these factors are considered the volumes of contaminated soil are likely to be very high. Removal of such large volumes of soil from a spill site or storage area will likely lead to extensive mechanical damage of the site, followed by extensive erosion. It is thus preferable that the soil not be removed from such contaminated sites, but that in situ treatment options be considered and/or developed.

8.3.12 Bio-remediation Industry and Research Institutions in South Africa

There are a number of South African institutions that conduct research and/or offer a commercial bioremediation service. Because there is a need for clean-up of especially oil spills and hydrocarbon polluted soils the institutions have focused their attention primarily on these pollutants. Only the Plant Protection Research Institute of the Agricultural Research Council and CSIR – Environmentek - Water Treatment and Remediation Technology groups have been involved with research in the field of bioremediation of pesticides. There are claims from some bioremediation service providers that their products will be able to bio-remediate pesticides. A company summary of major role players in the bioremediation business in South Africa is provided below.

Bio-systems South Africa

Company background

Bio-Systems SA offers scientific and engineering services to the municipal and industrial sectors for the improvement of effluent quality and reduction of pollution. The company was started in 1985 and has been providing environmental products and services to assist industry, municipal and commercial establishments to comply with regulations since inception. Bio-Systems Corporation Incorporated owns and operates the leading modern manufacturing facility for the production of powder and liquid bacterial cultures under stringent quality control procedures that meet the ISO/9002/1994 standard, and qualify them for registration with Lloyds as a Quality Company.

Company services

The company is a waste handling and bioremediation company utilizing specialist natural microbial products and innovative equipment, in accordance with environmental common sense and recognized hazardous waste regulatory disposal requirements.

Company Products

The biological products used are prepared from universal natural non-pathogenic Group 1 micro-organisms and do not use genetic modification in developing their bioremediation products. The Bio-systems products are internationally accredited for environmental compliance, carrying the approval of the United States EPA, the South African Government Departments and Local Provincial Authorities.

Bio-Systems have selected and adapted natural microbial cultures for the management of Liquid Effluent from Municipal, Agricultural, Industrial, Sewage Disposal Systems, Sewers, Odour control, oil soaked railway ballast, bunding aggregate and sand after spills, Management and maintenance of Grease Interceptor Traps (catering or mechanical).

The company makes extensive use of the bioremediation agent B350 in their bioremediation projects. B350 is designed to rapidly establish a biomass, in liquids or soils, which under suitable conditions, degrade pollutants quickly and efficiently. The product consists of a high concentration of Group 1 non-pathogenic microorganisms. Application protocols have been designed and extensively tested on a number of pollutants.

Pollutants considered

The South African commercial market for bioremediation technology to date has focused on hydrocarbons, with special reference to petrochemicals. To date the market for bioremediation of pesticides and pesticide associated waste has been nominal. Because of this the Bio-systems company have not been actively involved in pesticide bio-remediation. The company therefore currently specialises in bioremediation of Hydrocarbons in soil, chlorinated organic compounds as well as refinery and petro-chemical waste.

The pollutants that are known to be effectively degraded using their B350 product include diesel, kerosene and other petrochemicals, PCBs, PAHs as well as organochlorine pesticides.

MRO Product Management (Pty) Ltd

Company background

The company is a South African affiliate to the Entrech Australasia group. The company uses bioremediation to rid of polluted soils of contaminants and respond to contaminants spills. The company specialises in the bioremediation of PCBs, hydrocarbons, oil, grease, and sludge.

Company services

The company specializes in cleaning of hydrocarbon spill sites and bio-remediation of hydrocarbon polluted sites.

Company Products

MRO uses the Enretech-1 hydrocarbon biodegradation process. It is an extension of Enretech oil and fuel absorbents. The absorbents are made from recycled waste cellulose from the cotton. The absorbents are manufactured from certain portions of cotton waste, which allows for the use of micro-organisms that occur naturally on the cotton in the bioremediation process. The natural wicking action 'feeds' the contaminants through to the microbes where the contaminant is used as carbon source. The initial application of the product causes the encapsulation of the pollutants limiting it's leaching and inhibiting volatilisation of, for example, lighter fraction hydrocarbons.

The remediation process has been refined over several years of research and development, which has produced a complex and robust system that transfers easily to different soil types and climatic conditions. The application procedures are simple and non-labour intensive, making the process cost-effective. The micro-organisms used by the company have not undergone any genetic modification or bioengineering.

Pollutants considered

The South African commercial market for bioremediation technology to date has focussed on hydrocarbons, with special reference to petrochemicals. To date, the market for bioremediation of pesticides and pesticide associated waste has been nominal. Because of this the Bio-systems company have not been actively involved in pesticide bio-remediation. The company therefore currently specialises in bioremediation of Hydrocarbons in soil, chlorinated organic compounds as well as refinery and petro-chemical waste. The Enretech may also be applied for the bioremediation of organochlorine pesticides, a number of solvents and herbicides as well as vinyl chloride.

Experience has shown that where a soils is contaminated with up to 40,000 part per million of petroleum hydrocarbons, a single tilling of the absorbent-microbial blend may be sufficient for the bioremediation process to be completed within one to three months. Contamination levels between 40 000 to 150 000 ppm would take between two and six months depending on the type of hydrocarbons. Sludges containing in excess of 750 000 ppm contaminants can be remediated with regular tilling using clean soil or green waste compost to aerate the blend. At high levels of contamination, further additions of Enretech-1 or additional nitrogen sources may be necessary to achieve the fastest possible results.

Enretech-1 Environmental Remediation Technology - Pcb Case Study on Transformer Oil

Introduction

PCBs are a series of compounds which are sometimes found in transformer oil in the base of power transformers. PCBs are polychlorinated bi-phenyls, and research has proven that exposure to them is hazardous both to humans and to the environment. The American EPA has passed legislation governing the disposal of PCB's.

Each power transformer which contains PCBs in its oil is a potential environmental hazard. This case study, executed in Clinton, USA, will offer a cost-effective, and more importantly, environmentally safe solution to the removal and disposal of PCBs.

Procedure

Transformer oil containing 106ppm PCBs (34ppm of PCB 1242 and 72ppm of PCB 1260) was absorbed to saturation in Enretech Cellusorb (Enretech-2 Fibre) and blended with Enretech-1 until dry to the touch. The sample was wet thoroughly and maintained above 200 C at a 30% moisture level. Samples were taken every 30 days for a period of 240 days and sent to an independent chemical analysis firm for analytical determinations of TPH (Total Petroleum Hydrocarbons) and ppm (Parts Per Million) PCBs from within the mixture. Before the samples were taken, the contents of the test area were mixed thoroughly in an effort to distribute the contents evenly in order to gain a more accurate representation of the chemical constituents within the mixture.

Results

The results of this test clearly showed a reduction of PCB levels when treated with Enretech-1 and Enretech Cellusorb (E-2 Fibre).

PCB 1242 reached a level of 1.4 ppm at 90 days and non detectable in 120 days, while PCB 1260 reached 1.4 ppm in 210 days, and non-detectable in 240 days.

Conclusions

PCBs can be effectively bio-remediated using a mixture of Enretech-1 and Enretech Cellusorb (E-2 Fibre). The application of this type of remediation offers a cost-effective environmentally safe solution to the remediation of power transformer PCBs.

The experiment also proved the efficacy of Enretech E1 in bioremediating hydrocarbon products, like transformer oil. The TPH level reduced from 100% (1 million ppm) to 1.4% in 180 days, and to non-detectable in 240 days.

The following table gives the results obtained.

No. of days	% TPH	ppm PCB	
		1242	1260
	0	100	34
30	60	16	43,2
60	42	8,2	31,7
90	22	1,4	23
120	7,5	ND	15,8
150	4	ND	8,6
180	1,4	ND	4,3
210	0,3	ND	1,4
240	ND	ND	ND

CSIR – Environmentek - Water Treatment and Remediation Technology

The Water Treatment and Remediation Technology business area of CSIR – Environmentek is a research centre institution specializing in scientific research for water management in southern Africa. The focus area offers specialist consulting and research services that cover a full spectrum of physical, chemical and biological aspects of water, with expertise in groundwater, surface water and associated aquatic ecosystems. Services are offered to water resource and water quality managers, industry, water users and suppliers and regulatory bodies tasked with managing southern Africa's water resources.

Key services relevant to bioremediation include:

Groundwater resource management, Groundwater pollution assessment, Water treatment technologies, Water quality management, Decision support systems for cleaner production and waste management and aquatic chemistry and microbiology.

The activities of the Waste Treatment and Remediation Technology thrust (WTRT) are focused on the chemical, physical and biological treatment of industrial wastewater and the bioremediation of soil. The biological option to treat organics in an industrial effluent is an attractive alternative to the physico-chemical treatment due to its lower costs and efficient elimination of organic material. Both aerobic and anaerobic biological treatment protocols are used.

Aerobic wastewater treatment is used when the Chemical Oxygen Demand (COD) is lower than 2000 mg/l, whereas anaerobic treatment is used, when high COD concentrations are required. Two examples of treating wastewater aerobically, executed by the institution are the treatment of a high strength waste produced by a company manufacturing a product suitable to wash diamonds and the treatment of an effluent, produced by a carpet factory.

The anaerobic treatment system was applied to treat a high strength leachate where the so-called "in situ" bacteria were compared to the anaerobic bacteria.

WTRT have focussed their attention primarily on bioremediation of wastewaters, concentrating on hydrocarbons pollutants. They have also been involved with aspects of bioremediation with regard to:

- The herbicide 2,4-D in soil using a plasmid-mediated bio-augmentation system
- Diesel and other petrochemical bioremediation
- The use of a bio-indicators as indicators of Polycyclic Aromatic Hydrocarbons (PAHs) removal from soil
- Removal of hydrocarbons from mining property through bioremediation
- Post Harvest Decontamination of Pesticide treated soil
- Biological sulphate removal from mine water
- Bioremediation through land farming
- Soil remediation of oil and diesel-contaminated
- The use of alcohol-consuming micro-organisms can improve the quality of acid mine drainage

Agricultural Research Council – Plant Protection Research Institute

Bioremediation of herbicide polluted soils in South Africa – a case study

The Plant Protection Research Institute of the Agricultural Research Council is involved in research on the identification and isolation of suitable rhizosphere micro-organisms and plant species for their combined use in remediating herbicide polluted soils. There are sites located in South Africa that have been treated with particularly persistent herbicides for the purpose of controlling vegetation in these areas. The main aim of the herbicide application was to keep the area free of vegetation in order to aid in securing the facilities. To this end soil applied herbicides containing the active ingredients tebuthiuron, bromacil and ethidimuron were applied to the sites annually over a 13 year time period at rates of up to 8 kg/ha. The soils contained herbicides at concentrations ranging from 200 to 3000 µg/kg. The concentrations even at the lower end of the range were sufficient to keep the areas free of vegetation.

At some of the study sites, the weed control program almost completely eradicated vegetation. Bare soil was exposed to destructive actions of rain, wind and temperature, resulting in progressive erosion, and consequently, topsoil layer removal. At other sites, only limited plant populations were found, and where present consisted primarily of grasses and herbaceous weeds. It was clear that these sites were disturbed and that the lack of vegetation led to more severe damage in the areas.

The client opted for the development of an in situ phytoremediation system combined with the use of rhizosphere-associated micro-organisms to remove the contaminant herbicides from soils.

Background information regarding climatic conditions and herbicide use was gathered. Sites were mapped, and soil sampled for herbicide residue analysis and soil typing. Selected grasses were collected to test their herbicides resistance in a greenhouse experiment. Rhizosphere soil was screened for the presence of herbicide-degrading micro-organisms.

The purpose of this study was to investigate the potential of specific plant-bacteria combinations to phytoremediate herbicide polluted soils.

Several of the plant species growing in contaminated soils including *Salsola kali*, *Cynodon dactylon* and *Eragrostis curvula* were identified as herbicide tolerant plants for use in stabilising the soils. It was decided to make use of *Cynodon dactylon* in further studies, as this is a creeping grass species, which is ideally suited for soil stabilisation.

Bacterial cultures were prepared from the rhizosphere of grasses growing in herbicide-polluted soil. Results of bacterial counts indicated that vegetated soils contained 10-250 more bacteria than the bare soils. A selective enrichment technique was applied to the extract and bacteria that use ethidimuron and tebuthiuron as sole carbon sources were isolated.

During further trials it was shown that at least one of the isolated strains could be used to remove herbicides from contaminated soils.

Amphigro cc

The company AMPHIGRO CC research and development has succeeded in developing commercial inoculants for agriculture, horticulture and the rehabilitation industries in southern Africa. The company maintains a living culture collection and actively isolates new arbuscular mycorrhizal fungi (AMF) species for various purposes based on soil pH, P content and environmental factors such as climate and soil texture. The company encompasses 80% of mycorrhizologists countrywide and has collective experience of more than 50 years in the application and research of AMF. Company expertise includes various inoculum production methods including *in vitro* techniques, application of AMF for rehabilitation of goldmine slimes and coalmine and fly-ash tailings, various commercial agricultural applications and developmental rural agriculture. The collaborators and members of this company are from various institutions:

Mrs Beatrix Bouwman, School for Environmental Sciences and Development, Potchefstroom University for CHE, Dr Joanna Dames, Botany Department, Rhodes University, Dr Colin Straker, School of Molecular and Cell Biology, University of the Witwatersrand, Dr Robert Sinclair, Amphigro Pretoria Laboratory, Dr Alexander Valentine, Department of Botany, University of Stellenbosch, Private Bag X1, Matieland, South Africa.

Some of the limitations for the implementation of phytoremediation technologies are the contact between roots and pollutants, the growth of the roots and the toxic effects of pollutants (Leyval *et al* 2002).

The AMF can explore large volumes of soil on behalf of the symbionts, improve plant growth through better nutrition and alleviate toxicity of pollutants such as heavy metals. Metal tolerant AMF have been identified and isolated and experimental work has shown that AMF may be able to contribute to phytoextraction and phytostabilization of heavy metals and radionuclides depending on the plant – fungus partnership. There is currently not sufficient research data to enable us to make phytoremediation programmes with AMF a standard practice and especially the use of specially selected isolates for the purpose is paramount and an ongoing process.

The impact of AMF on PAH polluted soil is still uncertain, but the potential of plant establishment, survival, and degradation, would confer advantages that could justify the use of AMF inoculants when doing rehabilitation and revegetation. After fire, PAH is found naturally in soils and it is known that selection pressures are exerted on certain AMF groups such as the Glomales after fire, making the tolerance to PAH not improbable.

Arbuscular mycorrhizas are not only an aid in ecosystem remediation but can be considered a key indicator for soil pollution and soil quality because:

- mycorrhizal fungi are ubiquitous organisms
- many plants are highly dependant on mycorrhizas for their growth,
- they provide a direct link between the soil and the roots
- are involved in the transfer of elements including pollutants from soil to plants.

AMF should be considered as additional indicator organisms in the existing battery of bioassays. This is made easier due to the existence of commercial AMF inoculum now being available.

Microbial Solutions (Pty) Ltd

The company was founded in 1995 and is South Africa's oldest and most established player in the biological control and organic products markets. The company manufacture technologies, giving rise to high quality products having the ability to solve a wide range of problems when used alone or in strategic combinations.

All their products are certified as being 100% organic and are presented in highly concentrated formats for ease of use and reduction of costs. Field efficiency, positive environmental impact, user friendliness, and economy of costs are our constant goals. Using nature's own strengths Microbial Solutions creates a new safe, cost efficient world for all.

Whilst Microbial Solutions' focus is on agriculture, it also manufactures solutions for manure inoculation, water cleaning, algae control, septic tank and pit latrine management, and hydrocarbon-bioremediation - the breakdown of oil pollution using only natural means.

Microbial Solutions' product range currently encompasses living microbes, enzymes, and humic and fulvic acid preparations. In the pipeline are a host of new organic solutions.

8.3.13 Bio-remediation Research and Laboratory Capacity Requirements for Bio-remediation

Introduction

Analytical laboratory capacity is essential for the analyses and identification of obsolete stocks during inventory taking irrespective of the destruction technology envisaged. Similarly identification of unknown compounds and the determination of the extent of soil contamination and the proximity of the spill plumes will be required irrespective of destruction technology. However, in the case bio-remediation technology being applied, the residue levels must be known, thus a need for residue quantification exists. Furthermore, once the remediation process is in progress the extent and rate of remediation will have to be monitored through a detailed residue-monitoring programme, allowing for the verification of final clean up.

Analytical capacity is further required to identify historic storage sites for obsolete pesticide and POP stocks, but also to set priorities with regards to the order in which sites should be cleaned. It is obvious that the laboratory expertise required will not be limited to analytical capacity, but also expertise in the field of risk analysis and assessments.

Research and laboratory capacity will thus be required during the inventory phase (macro levels of pollutants), during the verification of decontaminated sites (micro concentration levels) and during specialist phases for the identification of unknowns and mixtures, finding polluted sites and carrying out impact and risk assessment.

Analytical Laboratory capacity in South Africa

In an exercise such as a clean-up through bio-remediation it is essential that data collected and reported be reliable and fully traceable. It is thus preferable that the results from such investigations are generated under an internationally accepted accreditation scheme. In South Africa the South African National Accreditation System (SANAS) is accredited and responsible for the compliance of accredited laboratories to OECD Good Laboratory Practise (GLP) guidelines. There are suitably accredited laboratories currently operating in South Africa.

In addition the expertise and capacity for conducting monitoring exercises and risk assessments are well established in the country. The links with research institutions throughout Africa is good and it is thus possible to form consortia to tackle the stockpiles problem throughout the continent with relative ease.

The equipment required for such exercises are all available in the country and trained operators are in place.

Vegetation and Microbiological laboratory and research capacity

An essential part of a bioremediation system is the capacity to isolate biodegrading micro-organisms as well as facilities to mass produce such strains. There is adequate capacity available in South Africa to perform both these functions. Capacity exists in the fields of nutrient fixing microbes, pollutant-degrading micro-organism as well as in the field of bio-technology.

A well-established network of plant breeding facilities is available in South Africa. There are institutions that specialise in the breeding and production of plant species specifically for their use in rehabilitation of disturbed sites.

8.3.14 Agricultural Research Council – Plant Protection Research Institute Plant Pathology and Microbiology Division

The Plant Pathology and Microbiology Division provides specialist research and diagnostic services on fungal, bacterial and viral plant pathogens with emphasis on those which affect crops. The division also specialises in research regarding advantageous micro-organisms and provide services to promote the exploitation of nitrogen fixation and mycorrhizal mediated nutrient uptake by crops of commercial as well developing agriculture. In addition, the division is involved in extraction and multiplication of micro-organisms capable of degrading soil pollutants.

8.3.15 Agricultural Research Council's Vegetable and Ornamental Plant Institute Biotechnology Division

The Division specialises in research on vegetables, ornamental plants, indigenous and grain crops. The Biotechnology Division offers expertise in research, development, and transfer of biotechnology products and services to resource poor and commercial farmers, plant breeders and other agricultural organisations. Research projects have been identified and implemented by the Biotechnology Division with the aim of developing new cultivars better suited to South African conditions.

The group has been actively involved in breeding herbicide resistant grass plants using specialised tissue culture techniques.

8.3.16 Possible other South African Role Players

Examples of institutions that could partake in such an exercise in South Africa are the Plant Protection Research Institute of the Agricultural Research Council, the Test House Laboratories of the South African Bureau of Standards, the electricity supply company ESKOM, the CSIR-Environmentek group, risk assessment consultancy companies such as Infotox as well as bioremediation service providers such as Bio-Systems, MRO Product Management and Prokon as well as Universities and other teaching institutions and non governmental organisations such as AfricaBio. The institutions mentioned above are all involved in research and development throughout the African continent.

8.3.17 Needs for Capacity Building, Technology Transfer and Small Business Development

Capacity building in Africa is required for implementation of bioremediation. Within the South African and African scenario there is a strong need for formation of consortia, which must include institutions of higher learning. It is important that not only is the continent rid of contaminants, but that capacity building becomes a priority. This goal can only be achieved through *in situ* destruction technology. Where pollutants are destroyed through incineration, there is no technology transfer and no capacity building as the pollutants are simply removed by "expert" non-African groups and incinerated. The bioremediation system however allows for capacity building and the development of small to medium scale businesses that could prevail and branch out into other spheres of environmental clean-up even after the stock piles have been removed. Africa needs to improve its capacity to manage pesticides, pollutants and POP's and require long term capacity to apply the technologies to solving pollutant problems in general, not only the POP's and pesticides stock piles currently.

8.3.18 Advantages and Disadvantages of Applying Bioremediation Technology

Disadvantages

- The main disadvantage of implementation of bioremediation systems as a destruction technology lies in the longer time required for clean-up. The sites to be addressed are likely to contain a mixture of contaminants. A single micro-organisms system is not likely to degrade all the pollutants components. It will likely be necessary to isolate and culture stocks of micro-organisms for some pollutants. There is thus likely to be some development phase required prior to implementation.
- The implementation of such a project will require training of local personnel to enable them to apply all required aspects of the technology.
- Bioremediation requires a longer term monitoring action to be implemented through which the pollution status must be determined
- Some compounds may not be prone to microbial degradation and may require an additional chemical degradation step or bio-augmentation step.

Advantages

- The need for trained personnel at local level allows for potential development of small to medium scale enterprises, which could aid in job creation in especially rural areas. The technology required to multiply micro-organisms and plants may be ideally suited for small business development. In addition the technology developed will remain in the country and continent.
- The technology has a high potential for technology transfer
- The introduction of plants as part of the phytoremediation part of the system, allows for the system to be stabilised. There is no need for the removal of large volumes of soil, which will lead to extensive erosion.
- Non-disturbance of the soil will insure soil integrity and fertility remains intact.
- The implementation of a bioremediation system will require labour, which can be provided through training of locals.

The implementation of the system is likely to be more cost effective than incineration, especially where soil contamination is addressed

8.3.19 Biobeds

Although it is recognized that pesticides have an important role to play in modern agriculture, it is equally well accepted that these agrochemicals may contaminate the environment. Public pressure

against the presence of agrochemicals in potable waters has reached such proportions, in developed countries in particular, that the continued use of these indispensable crop production tools are threatened. Concern about pollution caused by agrochemicals in developing countries like South Africa is fast catching up with the negative perceptions in the developed world, and this awareness is reflected in green and white papers on environmental protection.

Agriculture would be wise to be pro-active in the development of strategies for curbing pesticide entry into sensitive compartments of the environment. Water resources and the atmosphere are arguably the two most sensitive parts of the environment. The two broad categories of pesticide pollution are point sources and non-point or diffuse sources. The former category is represented by concentrated, accidental spillages, as well as spillages during filling and washing of application equipment. Diffuse pollution, on the other hand, is represented by application of agrochemicals on crop fields. Considering the volumes of pesticides used it is actually surprising that so few have been detected outside the target areas (crop fields), and if they are, the concentrations are usually very low.

Significance of pesticide persistence for groundwater contamination

Key criteria of pesticide performance are: activity, persistence and crop safety. Persistence describes the residence time of a pesticide in the environment. Logically, the longer the persistence of a compound, the greater the potential for it to contaminate air, water or soil. Persistence has an ambiguous nature; it should be long enough to provide adequate pest control for as long as possible in the season of application, but it should not be so long that it carries over to sensitive crops (particularly herbicides), nor is it supposed to reach sensitive off-target areas such as surface- and groundwater. Excessive persistence of pesticides presents the single greatest risk to water resources. Other determining factors are rates of application, and factors promoting the movement of pesticides either vertically (leaching in soil) or horizontally (surface run-off). The amount of overhead water (rain or irrigation water), and the strength by which pesticides are retained on soil particles, determine the mobility of pesticides in and on soils.

The persistence of pesticides is the overriding factor in the determination of their potential to reach groundwater sources in detectable amounts. For example, a highly leachable compound that has a very short persistence is unlikely to pose a threat to groundwater, provided the water table is not very high. Even if such a compound does reach the groundwater, its limited persistence will probably render it harmless quickly. On the other hand, a highly persistent compound with low mobility in soil, may eventually accumulate to unacceptable levels in groundwater if given enough time. Thus, both persistence and adsorption/leaching characteristics of pesticides must be considered in assessments of their potential to contaminate groundwater. No pesticide has an inherent persistence (expressed as half-life, i.e. the time required for the original concentration to be halved in a particular situation). Due to the influences of many factors and their interactions, the persistence of a compound can vary considerably from soil to soil, from one site to another, from year to year, and from season to season. Leaching behaviour can also not be expected to be constant for a particular compound.

8.3.20 Pesticide losses and risk assessment

The fate of a pesticide applied to soil is governed to a large extent by its adsorption on soil colloids, desorption of the adsorbed fraction, its degradation and persistence. Part of the amount of a pesticide that is sprayed onto plants can be expected to eventually reach the soil through wash-off by rain, or simply, through the spray solution missing the target (the plant) and landing directly on the soil surface. Pesticides are lost from soil in many ways. By definition, loss infers that the chemical is transformed into by-products, which are further transformed, and that the parent molecule and its by-products are also transferred to other places and transformed until none exists, or in most cases none is detectable. Hence the many processes involved in the loss of pesticides are categorized as transfer processes or transformation processes. Some of the more important avenues of loss are chemical

degradation, leaching, microbial degradation, uptake by living organisms, photo(light) decomposition and volatilization.

Prevention of groundwater contamination from a point-source in agriculture

It is possible to permanently trap and to inactivate pesticides in a controlled way in order to prevent, or at least, reduce water pollution. The concept involves containment of pesticides in a relatively small compartment of the environment that can be managed in such a way that point-source contamination, which may result from spillages during filling and rinsing of spray equipment, is avoided. As mentioned earlier, the inactivation of pesticides may be temporary through reversible retention (sorption) on adsorptive surfaces, or permanent due to degradation by chemical or biological processes. The idea is to maximize the rates of degradation processes in order to permanently neutralize potentially hazardous pesticides.

A site large enough to accommodate all sizes of pesticide application apparatus available on a farm needs to be selected (see illustration). A pit at least 1.5 m deep and of adequate surface area must be prepared. Organic matter that is either already well-composted, or have the ability to compost readily is used to fill the pit. The organic matter is compacted well enough that the weight of a tractor and/or sprayer can be supported. The organic matter in the system has a dual purpose, it has high pesticide retention capacity, thus preventing leaching, and harbour micro-organisms that actually feed on most organic pesticides. The organic matter is covered with a layer of gravel that forms the working surface on which the tractor/sprayer is parked during filling/cleaning operations.

The environment inside the trap system should ideally be acidic so as to promote chemical degradation of pesticides, and at the same time be conducive to high microbial activity. Liming should be avoided since many of the most persistent pesticides are structurally stable under neutral and moderately alkaline conditions, whether in water or in soil. The internal environment should be constantly moist in order that microbial activity and chemical degradation processes proceed at optimal levels. Both types of degradative processes occur in water medium, and will be slowed down or stopped if water content drops below critical levels.

A roof, either a lean-to or a carport, is strongly recommended to prevent excessive flooding of traps. Intense rain could fill the pit and cause trapped pesticides to escape in overflow. In addition to roofing, pits should ideally be sunk on raised ground. The water content of the organic matter should be checked regularly and adjusted by addition of water. The organic matter must not be allowed to dry out completely.

Over time the surface will subside. The system can be rejuvenated from time to time by addition of organic matter. Organic matter must not be removed from traps, and never be used in crop production. If the organic matter in traps needs to be destroyed for some unforeseen reason, burning is advised, since it should destroy all active pesticides as well. The PSST system is not designed to be a hazardous waste dump site, and all pollution measures that may be prescribed by the manufacturer of a pesticide product still need to be followed judiciously.

9. FIRST ORDER ESTIMATES OF NON-POINT SOURCE POLLUTION FROM AGRICULTURAL PESTICIDES

The 1st order estimates from NPS agricultural pesticide pollution is discussed in this section.

9.1 PESTICIDE USAGE IN SA

The data for following main pesticide usage figures were obtained from CropLife SA Pty (Ltd), Midrand, Johannesburg. The data is supplied as the volume of active ingredient (a.i.) sold per main pesticide group.

The major pesticide sales for the major pesticide classes are shown in Figure 9.1 to Figure 9.8.

9.1.1 Herbicides

Data shows that herbicide sales have increased from sales in 1994 to sales in 2000 (Figure 9.1). The largest sales over this period (1994 - 2000) were recorded for triazines and acetanilides, which have remained relatively constant whereas there was a steady proportional increase organophosphorous herbicide sales (Figure 9.2). Total herbicide sales volumes recorded ranged from 7500 tonnes to more than 11000 tonnes per year (Figure 9.1).

9.1.2 Fungicides

Data shows that fungicide sales have increased from sales in 1994 to sales in 2000 (Figure 9.3). The largest sales over this period (1994 - 2000) were recorded for inorganic-organometallic and carbamate-dithiocarbamate, which have increased relatively consistently whereas there was also a proportional increase in triazine fungicide sales in 2000 (Figure 9.4). Total fungicide sales volumes recorded increased from 3500 tonnes to more than 9000 tonnes per year (Figure 9.3).

9.1.3 Insecticides

Data shows that total insecticide sales have increased from sales in 1994 to sales in 2000 (Figure 9.5). The largest sales over this period (1994 - 2000) were recorded for hydrocarbons and organophosphates, which have increased over this period, whereas carbamate sales have remained relatively consistent (Figure 9.6). Total insecticide sales volumes recorded ranged from 3500 tonnes to more than 5000 tonnes per year (Figure 9.5).

9.1.4 Nematicides

There was an increase in nematicide sales over the recorded period from 1994 to 2000 (Figure 9.7). The largest recorded increase in nematicide sales occurred in the organochlorine class (Figure 9.8). Total nematicide sales volumes recorded ranged from 800 tonnes to more than 2000 tonnes per year (Figure 9.7).

9.1.5 Plant Growth Regulators (PGR)

There was a consistent increase in PGR sales recorded over the 1994 to 2000 period (Figure 9.9). The main source of increased PGR sales was aliphatic amides, unclassified, phenolic compounds and organophosphorous PGRs (Figure 9.10). Total PGR sales volumes recorded ranged from 1200 tonnes to more than 2300 tonnes per year (Figure 9.9).

9.1.6 Seed Dressings (SD)

A decrease was recorded in total SD sales over the 1994 to 2000 period (Figure 9.11). The volumes decreased from 150 tonnes per year to 38 tonnes per year (Figure 9.11). The largest total recorded sales occurred in the dithiocarbamates and benzimidazoles (Figure 9.12). Total recorded sales for the benzimidazoles and unclassified (various compound types) decreased to almost zero levels in 2000 (Figure 9.12).

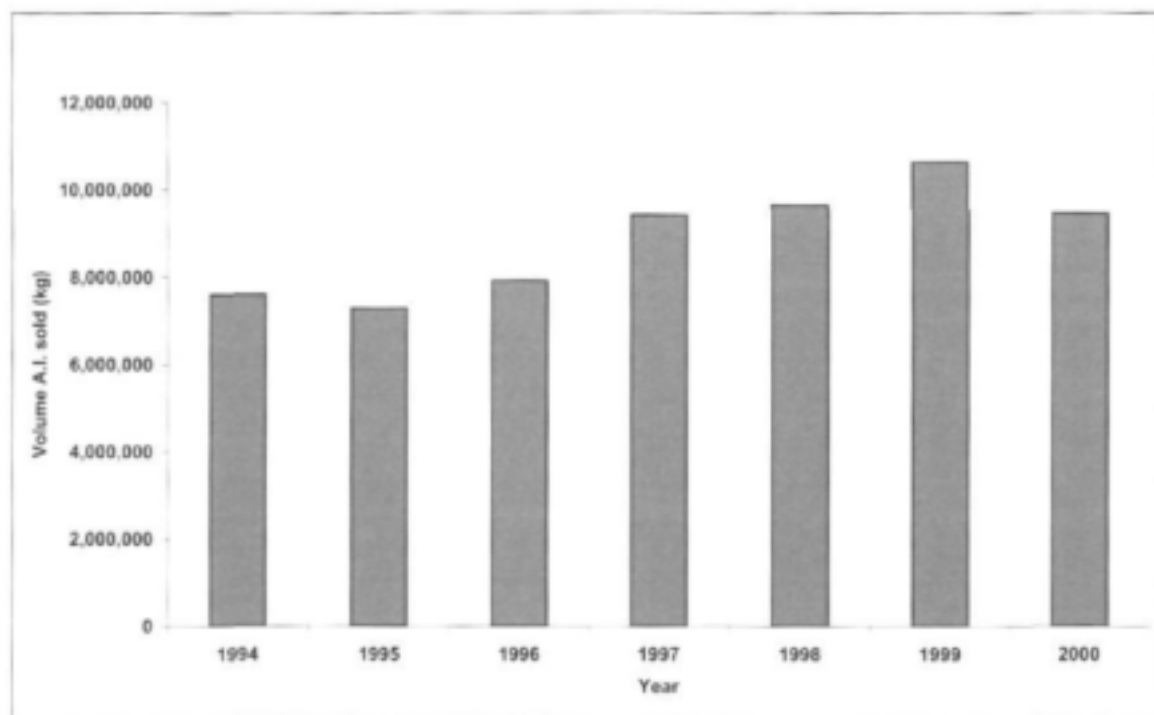


Figure 9-1 Herbicide sales recorded from 1994 to 2000 in SA

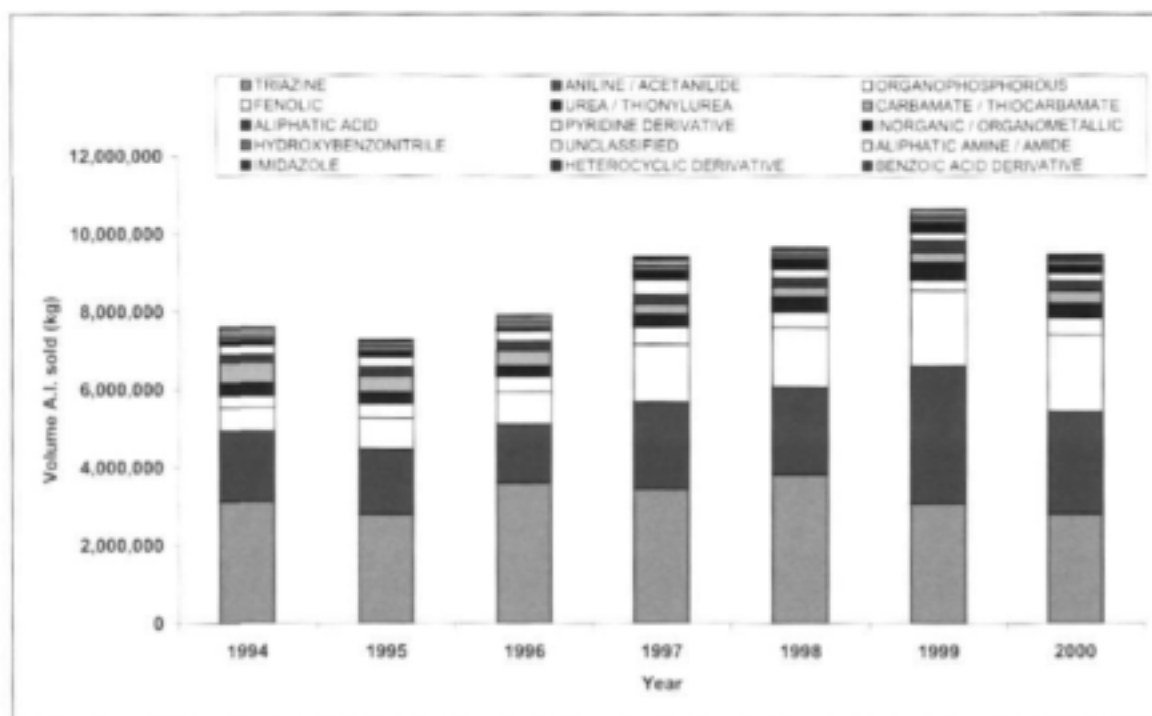


Figure 9-2 Different herbicide class sales recorded in SA from 1994 to 2000

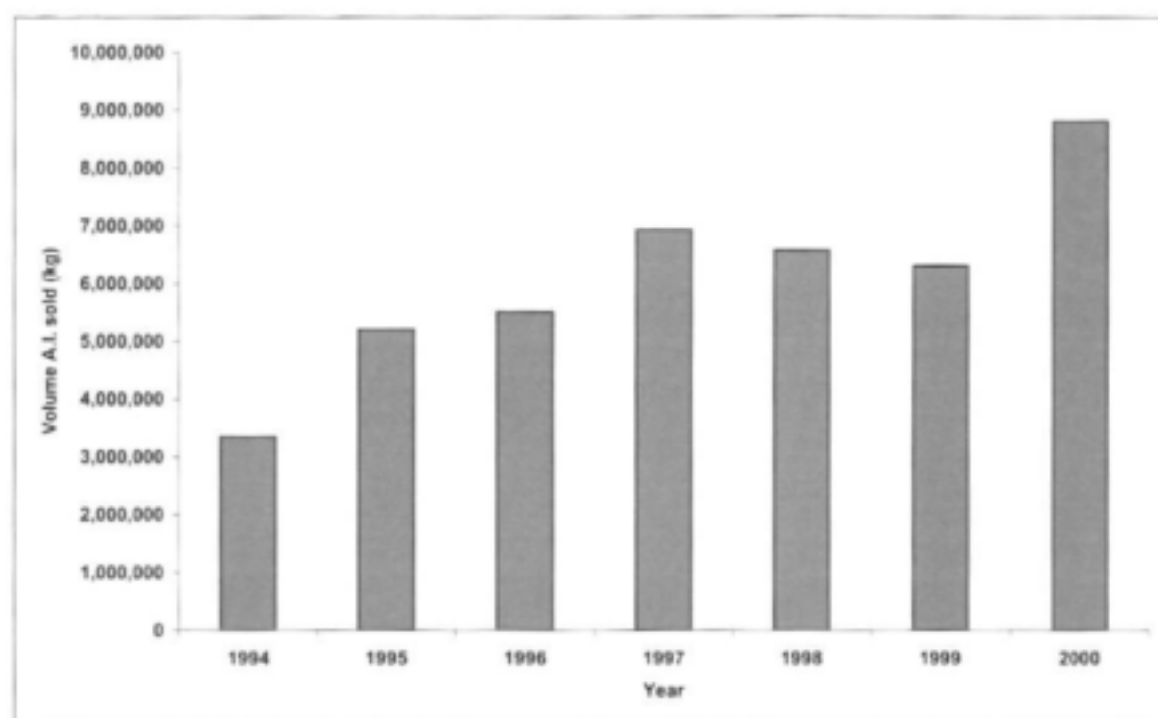


Figure 9-3 Total fungicide sales recorded from 1994 to 2000 in SA

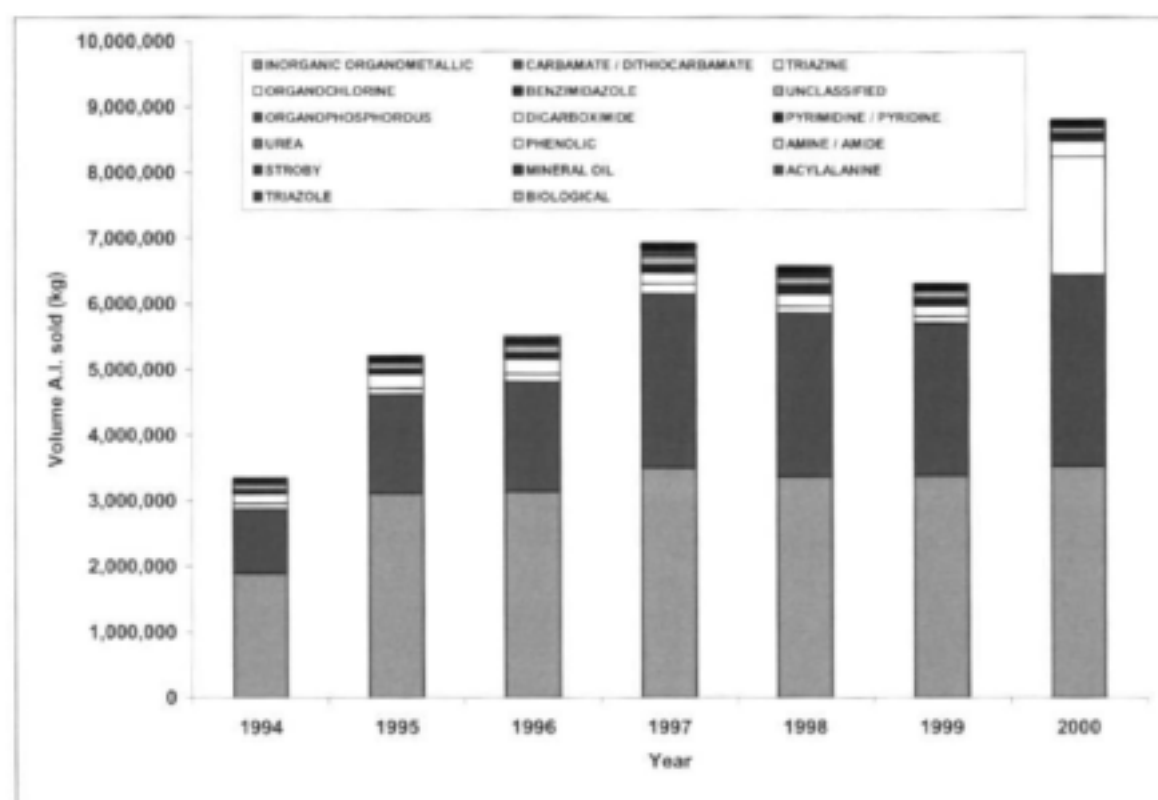


Figure 9-4 Different fungicide class sales recorded in SA from 1994 to 2000

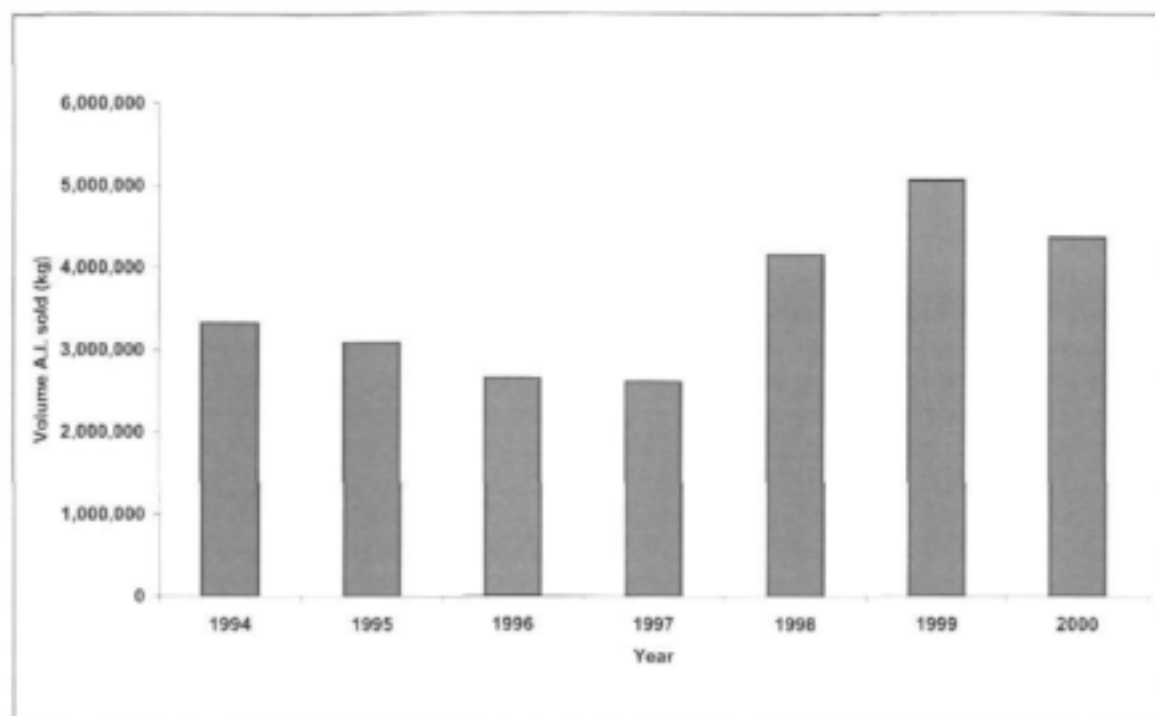


Figure 9-5 Total insecticide sales recorded from 1994 to 2000 in SA

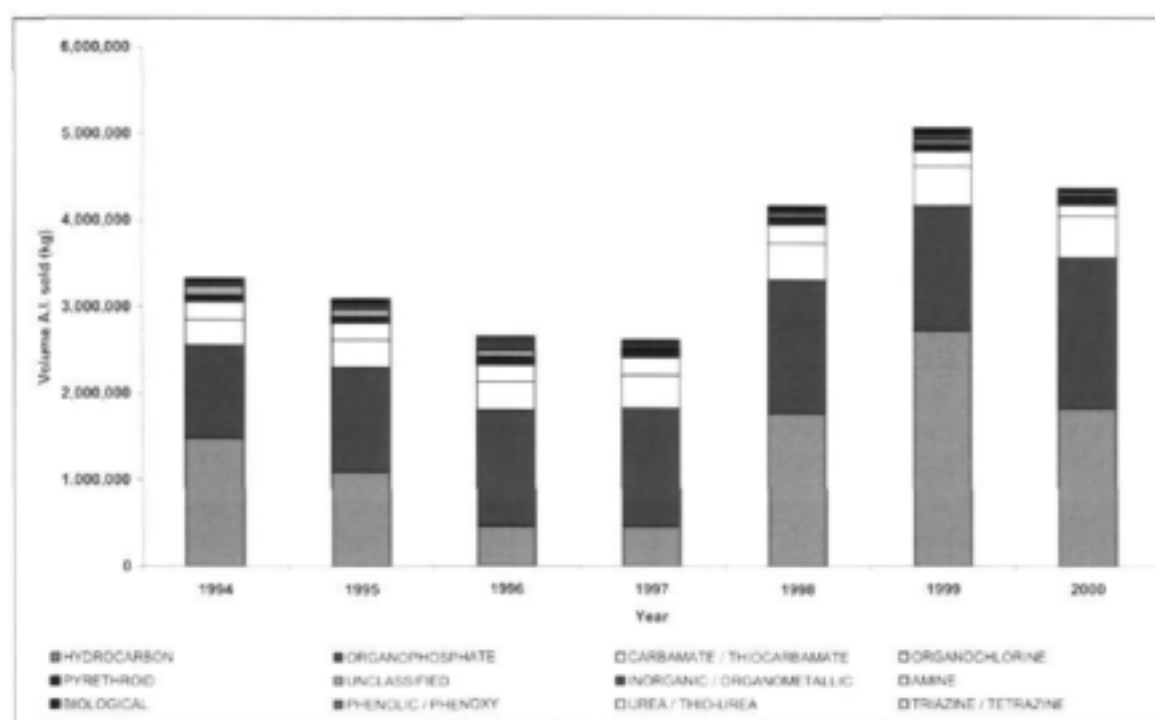


Figure 9-6 Different insecticide class sales recorded in SA from 1994 to 2000

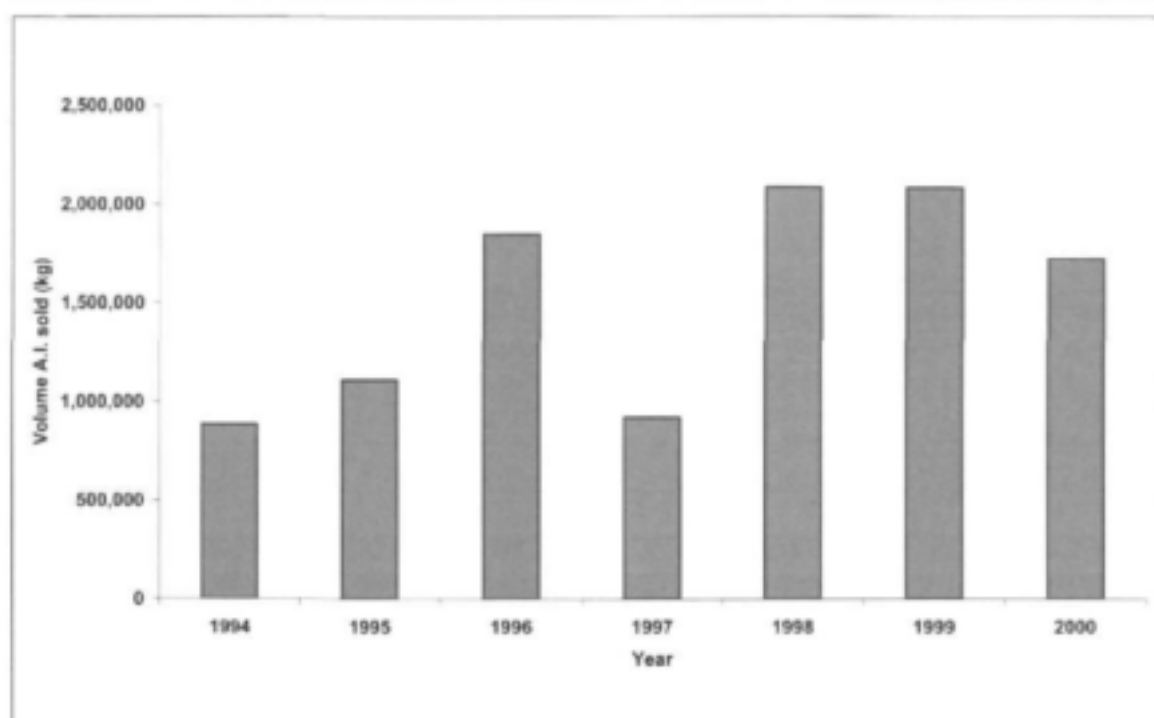


Figure 9-7 Total nematicide sales recorded from 1994 to 2000 in SA

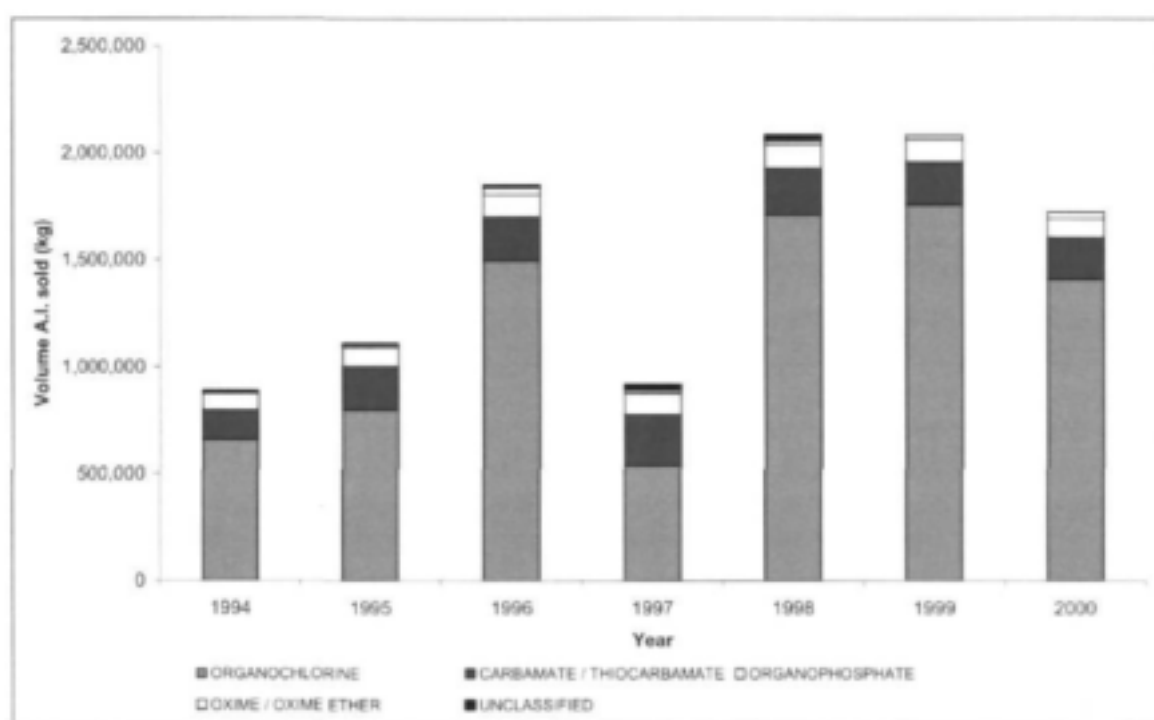


Figure 9-8 Different nematicide class sales recorded in SA from 1994 to 2000

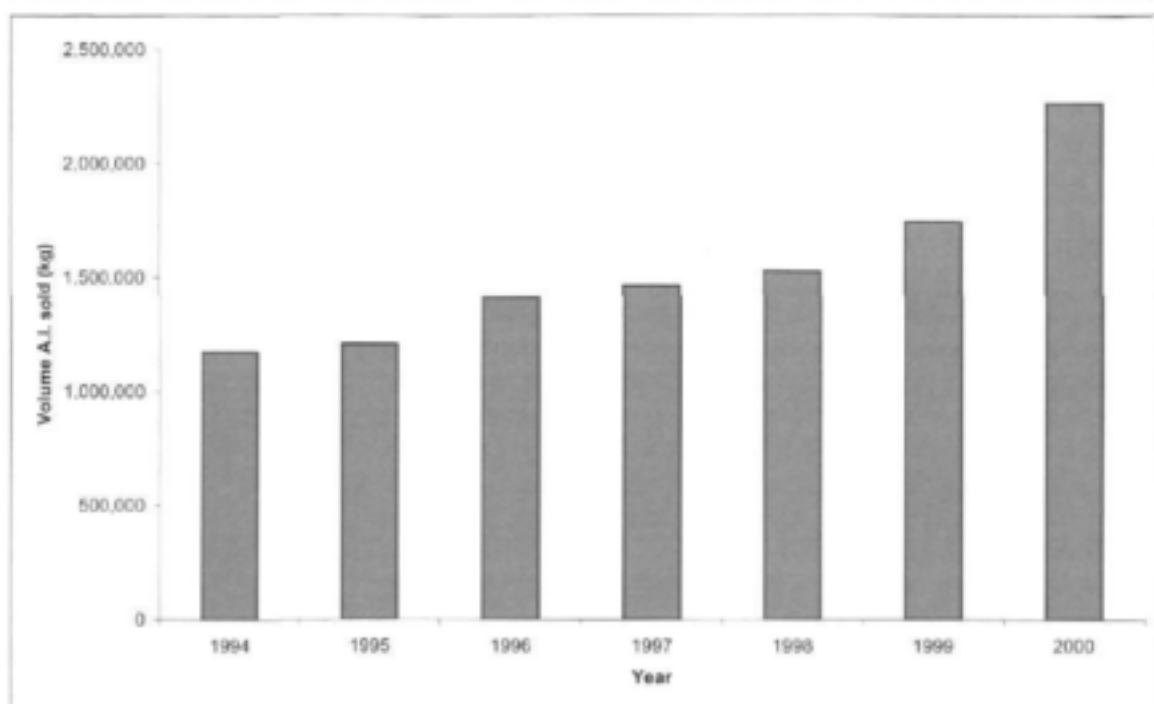


Figure 9-9 Total plant growth regulators sales recorded from 1994 to 2000 in SA

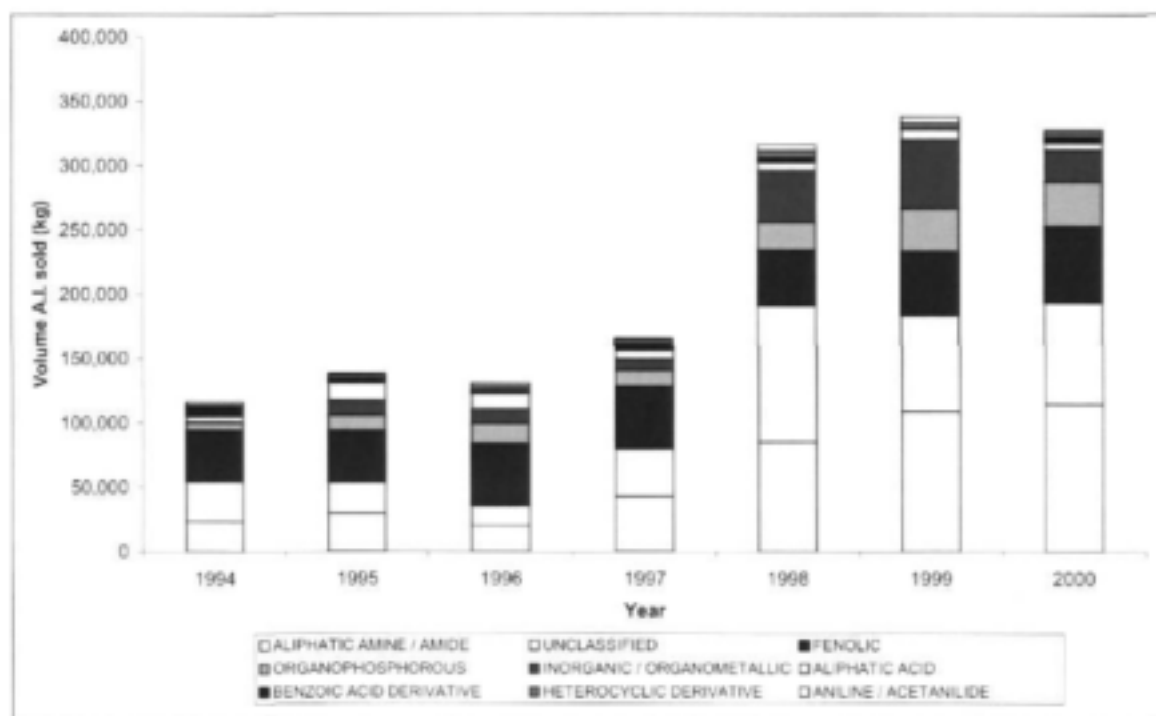


Figure 9-10 Different plant growth regulators class sales recorded in SA from 1994 to 2000

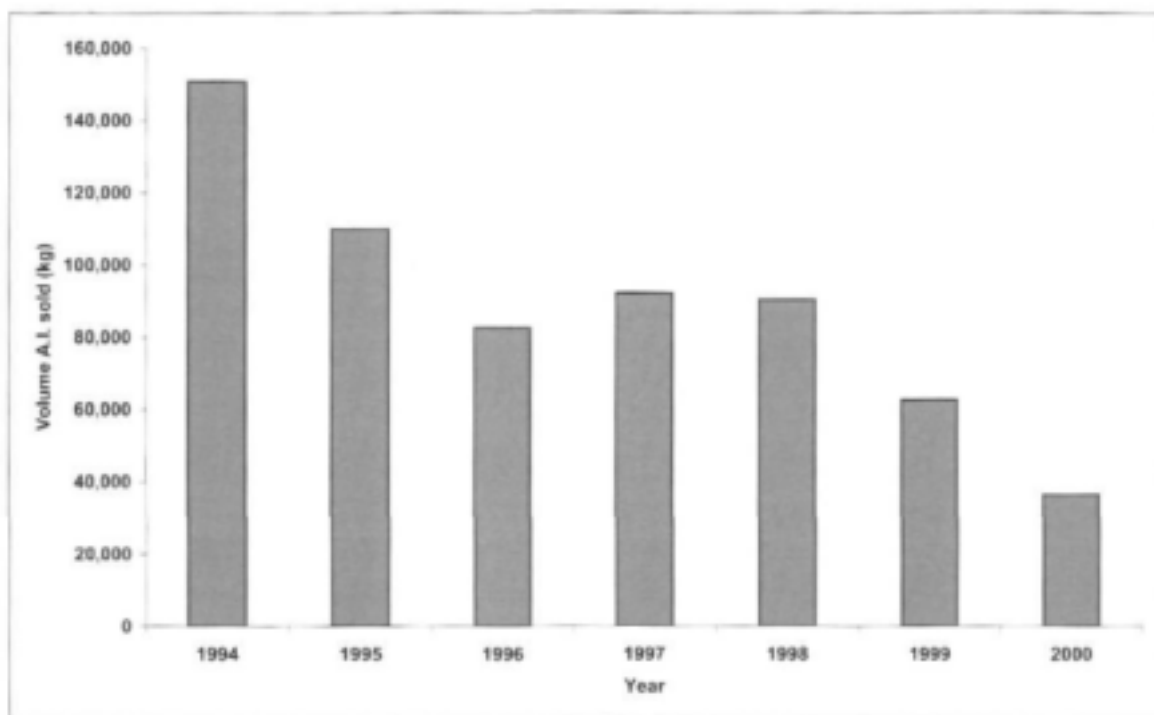


Figure 9-11 Total seed dressing sales recorded from 1994 to 2000 in SA

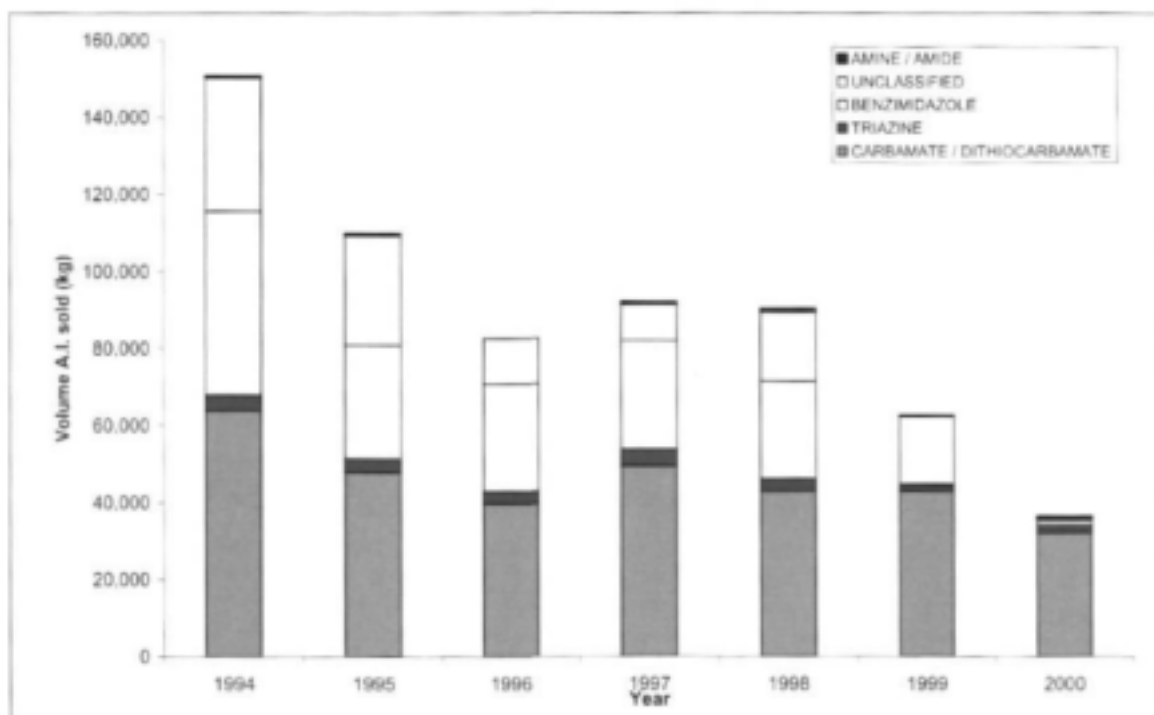


Figure 9-12 Different seed dressing class sales recorded in SA from 1994 to 2000

9.2 PESTICIDE USAGE IN SA

The ground water ubiquity score or GUS index is generally used to evaluate the leaching potential (soil mobility risk) of pesticides (ref. section 6 of this report). The GUS index is currently used as a 1st tier pesticide registration screen by the US EPA. This system does not form part of the South African pesticide registration process performed by the Department of Agriculture (DoA).

As stated previously in this report, the GUS index relies on the K_{oc} and determined half life DT_{50} of the pesticide in a specific soil type. This information is in most cases not available for South African soils and local conditions.

Herbicides are by far the most used pesticide class in SA, based on annual sales figures (Figure 9.1). Within the herbicides, triazines are by far the most widely used herbicide class of molecules (Figure 9.2). It is also this class (triazines) that are most likely to leach (based on the GUS index) and pose risk the general environment. Atrazine (one of the triazines) is a recognised endocrine disrupter (EDC) and is widely used within South Africa. Previous studies by the ARC-PPRI (Van Dyk, Pick, 1994) have shown widespread leaching of atrazine in groundwater. It is therefore most likely that endocrine disruptive effects in humans will become more common in future, especially in intensive farming regions and districts within South Africa.

HERBICIDES (GUS index) GUS threshold limit= 2.8

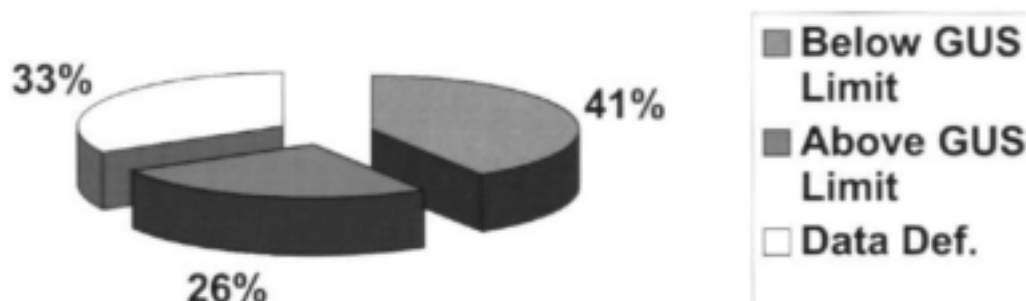


Figure 9-13 The GUS index evaluation of herbicide active ingredients, above and below the GUS threshold limit of 2.8

A substantial portion (26%) of the herbicide active ingredients that were evaluated according to the GUS index, fall above the GUS threshold value (2.8). It is therefore possible that many more herbicides may pose a soil mobility threat to the environment.

9.3 KNOWLEDGE GAPS

There remains a severe lack of adequate monitoring data for pesticides in the environment.

In order to make even reasonable predictions of NPS pesticide pollution, accurate field data is vital in order to validate and calibrate existing pesticide models and those in development.

Studies conducted by Dubrowski (2003) shows the enormous benefit in utilising actual field data in refining pesticide run-off models.

The model evaluation studies by Meinhardt (2003) has shown the enormous errors that may exist in models developed elsewhere in the world when applied to local southern African conditions.

9.4 FUTURE RESEARCH RECOMMENDATIONS

The following recommendations emanated from this report:

- A mechanistic approach is required to accurately predict pesticide runoff.
- Modelling of volatilisation needs to be validated in field trials.
- Pesticide-specific sorption coefficients need to be determined in laboratory experiments in order to accurately describe the adsorption/desorption process.
- Models including a sound plant water uptake subroutine are preferred in order to accurately simulate pesticide uptake by plants.
- The convection-dispersion approach has a more scientifically sound background, but it requires more inputs compared to convection approaches, in particular if processes like volatilisation, plant uptake and biochemical decay are solved simultaneously. In general, the level of detail needed to model solute transport and pesticide leaching is uncertain and case specific. Selection of the model or approach should therefore be based on the objectives that one wants to achieve.
- There is need to develop and adapt existing models to simulate preferential flow.
- There is also need to develop and adapt existing models to simulate fluxes and processes occurring in the intermediate vadose zone in order to allow for extrapolation from field to catchment scale.
- The factors controlling the chemical and microbial degradation rate of pesticides (soil properties, weather conditions, properties of the specific pesticide etc.) need to be investigated. This can be done under laboratory conditions, but data need then to be extrapolated to field conditions.
- Photo-decomposition at plant and soil surfaces needs to be validated in field trials, as there is evidence that it can contribute considerably to the pesticide balance.

10. BIBLIOGRAPHY USED IN THIS REPORT

Overview of the problem RSA

Bennett, E R, Hahn, C, Dabrowski, J M, Thiere, G, Peall, S K and Schulz, R. 2003. *Fate of aqueous-phase azinphos-methyl in a flow-through wetland in South Africa following a spray drift event*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Bollmohr, S and Schulz, R. 2003. *Seasonal changes in macroinvertebrate populations in a Western Cape River (South Africa) receiving non-point-source organophosphate pollution*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Bouwman, H, Vosloo, R, Maboeta, M and Bester, B. 2003. *An assessment of 30 years of environmental organochlorine residue data from southern Africa*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Dabrowski, J M and Schulz, R. 2003a. *Predicted and measured levels of azinophosmethyl in the Lourens River, South Africa: Comparison of runoff and spray drift*. Environ. Toxicol. Chem. 22(3) 494-500.

Dabrowski, J M and Schulz, R. 2003b. *Predicted and measured levels of azinophosmethyl in the Lourens River, South Africa: Comparison of runoff and spray drift*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Dabrowski, J M, Peall, S K C, Van Niekerk, A, Reinecke, A J, Day, J A and Schulz, R. 2002. *Predicting runoff-induced pesticide input in agricultural sub-catchment surface waters: linking catchment variables and contamination*. Water Research 36 4975-4984.

Dabrowski, J M, Peall, S K C and Schulz, R. 2003a. *Predicting runoff induced pesticide input in agricultural sub-catchment surface waters: Linking catchment variables and contamination*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Dabrowski, J M, Peall, S K C, Reinecke, A J, Liess, M and Schulz, R. 2003b. *Runoff-related pesticide input into the Lourens River, South Africa: Basic data for exposure assessment and risk mitigation at the catchment scale*. Water, Air and Soil Pollution 135 265-283.

Dalvie, M A, Cairncross, E, Solomon, A and London, L. 2003. *Contamination of rural surface and groundwater by endosulfan in farming areas of the Western Cape, South Africa*. Environmental Health: A Global Access Science Source 2003, 2:1.

Davies, H. 1997. *An assessment of the suitability of a series of Western Cape farm dams as water bird habitats*. MSc (Conservation Biology) thesis, Zoology Department, University of Cape Town, Cape Town.

Du Preez, L H, Carr, J A, Giesy, J P, Gross, T S, Hecker, M H, Jooste, A M, Kendall, R J, Smith, E E, Solomon, K R and Van der Kraak, G. 2003. *Exposure characterization and responses to field exposures of *Xenopus laevis* to atrazine and related triazines in South African corn growing regions*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Fatoki, O S and Awofulu, R O. 2003. *Methods for selective determination of persistent organochlorine pesticide residues in water and sediments by capillary GC-ECD*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Grange, A H, Papo, M T, Mathebula, S and Sovocool, G W. 2003. *Identification of compounds in South African stream samples using Ion Composition Elucidation (ICE)*. Conf. of the American Society of Mass Spectrometry, June 2003.

Heath, R G, Du Preez, H H and Genghe, B. 2003. *Human health risk assessment protocol development of consuming fish from a South African lowveld river*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

IGS. 2003. *Quantification of the impacts of groundwater pollution in urban catchments*. Progress Report No. K5/1326/1, Water Research Commission, Pretoria, South Africa.

Leslie, G. W. and Baxter, C. 2003. *Preliminary assessment of the effect of alpha-cypermethrin on arthropods associated with sugarcane fields*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

London, L and Myers, J. 1995a. *General patterns of agrichemical usage in the southern region of South Africa*. South African J. Sci. 91 509-514.

London, L and Myers, J. 1995b. *Agrichemical usage patterns and workplace exposure in the major farming sectors in the southern region of South Africa*. South African J. Sci. 91 515-522.

London, L, Dalvie, M A, Cairncross, E and Solomons, A. 2000. *The quality of surface and groundwater in the rural Western Cape with regard to pesticides*. Water Research Commission Rep. No. 795/1/00.

McGregor, F. 1999. *The mobility of endosulfan and chlorpyrifos in the soils of the Hex river valley*. MSc thesis, Dept. of Geological Sciences, University of Cape Town, Cape Town, South Africa.

Naidoo, V and Buckley, C A. 2003. *Survey of pesticide wastes in South Africa and review of treatment options*. Water Research Commission Rep. No. 1128/1/03, Pretoria, South Africa.

Papo, T and Mathebula, S. 2003. *Investigation of the occurrence of the Diazonin in the surface water of an urban river: KaalSpruit during the low flow periods with GC-MS*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Schulz, R and Dabrowski, J M. 2003. *Combined effects of predatory fish and sub-lethal pesticide contamination on the behaviour and mortality of mayfly nymphs*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Schulz, R, Peall, S K C, Dabrowski, J M and Reinecke, A J. 2001. *Current-use insecticides, phosphates and suspended solids in the Lourens River, Western Cape, during the first rainfall event of the wet season*. Water SA 27(1) 65-70.

Sereda, B L and Meinhardt, H R. 2003. *Insecticides in the South African water environment of the KwaZulu-Natal malaria endemic area*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Snyman, R G, Reinecke, A J and Reinecke, S A. 2003. *Field application of a lysosomal assay as biomarker of copper oxychloride exposure, in the snail Helix aspersa*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Solomons, A, Dalvie, M A, Cairncross, E and London, L. 2003. *Contamination of rural surface and groundwater by endosulfan in farming areas of the Western Cape, South Africa*. Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Van der Walt, E and Badenhorst, S. 2003. Environmental fate of fenthion, an avicide used for the control of the weaver bird: Red-billed quelea (*Quelea quelea* L.), in thorn bush ecosystems in South Africa). Joint European-Southern African International Conference on Pesticides in non-target agricultural environments – Environmental and economic implications, 21-23 January 2003, University of Cape Town, Cape Town, South Africa.

Weaver, J M C. 1993. A preliminary survey of pesticide levels in groundwater from a selected area of intensive agriculture in the Western Cape. Water Research Commission Rep. No. 268/1/93, Pretoria, South Africa.

Weaver, J and Reinhardt, C F. 1999. Chapter 8: Ground water contamination by pesticides, in: Conrad, J E, Colvin, C, Sililo, O, Görgens, A, Weaver, J and Reinhardt, C F. 1999. Assessment of the impact of agricultural practices on the quality of groundwater resources in South Africa, Water Research Commission no. 641/1/99, Pretoria South Africa.

Modelling soil processes

Aden, K and Diekkruger, B. 2000. Modeling pesticide dynamics of four different sites using the model system SIMULAT. *Agric. Water Manage.* 44 337-355.

Aller, L T, Lehr, J H and Petty, R J. 1987. DRASTIC: A standardized system for evaluating groundwater pollution potential using hydrogeological settings. EPA 600/2-87/035, US Environmental Protection Agency, Ada, OK, USA.

Armstrong, A C, Matthews, A M, Portwood, A M, Leeds-Harrison, P B and Jarvis, N J. 2000. CRACK-NP: A pesticide leaching model for cracking clay soils. *Agric. Water Manage.* 44(1-3) 189-199.

Bavcon, M. 2003. Investigations of the determination and transformations of diazinon and malathion under environmental conditions using gas chromatography coupled with a flame ionization detector. *Chemosphere* 50(5) 505-597.

Bergstrom, L F and Jarvis, N J. 1994. Evaluation and comparison of pesticide leaching models for registration purposes. *J. Environ. Sci. Health* 29 1061-1072.

Boesten, J J T I. 1999. From laboratory to field: uses and limitations of pesticide behaviour models for the soil/plant system. *Weed Res.* 40 123-138.

Boesten, J J T I and Gottesburren, B. 2000. Testing PESTLA by two modellers for bentazone and ethoprophos in a sandy soil. *Agric. Water Manage.* 44

Bomberger, D C, Gwinn, J L, Mabey, W R, Tuse, D and Chou, T W. 1983. Environmental fate and transport at the terrestrial-atmospheric interface. In: Swann R and Eschenroeder A (eds.) *Fate of chemicals in the environment – Compartmental and multi-media models for predictions*. ACS Symp. 225, 41-65.

Bonazountas, M. 1983. Soil and ground water fate modeling. In: Swann R and Eschenroeder A (eds.) *Fate of chemicals in the environment – Compartmental and multi-media models for predictions*. ACS Symp. 225, 197-214.

- Borchers, U, Peters, B and Overath, H. 1995. Limits and possibilities of using computer models to predict fate and behaviour of pesticides in soil. In: Pesticide movement to water, The British Crop Protection Council Monograph No. 62, 281-286.
- Bromilow, R H, Evans, A A and Nicholls, P H. 1999. Factors affecting degradation rates of five triazole fungicides in two soil types. 2. Field studies. *Pesticide Sci.* 55(12) 1135-1142.
- Cambell, G S and Norman, J M. 1998. An introduction to environmental biophysics. 2 ed. Springer, New York.
- Cohen, S Z, Wauchope, R D, Klein, A W, Eadsforth, C V and Graney, R. 1995. Off-site transport of pesticides in water: Mathematical models of pesticide leaching and runoff. *J Pure and Appl. Chem.* 67(12) 2109-2148.
- Cote, C M, Bristow, K L and Ross, P J. 1999. Quantifying the influence of intra-aggregate concentration gradients on solute transport. *Soil Sci. Soc. Am. J.* 63 759-767.
- Diekkruger, B, Sondgerath, D, Kersebaum, K C and McVoy, C W. 1995. Validity of agro ecosystems models: A comparison of results of different models applied to the same data set. *Ecol. Modelling* 81 3-29.
- Dust, M, Baran, N, Errera, G, Hutson, J L, Mouvet, C, Schafer, H, Vereecken, H and Walker, A. 2000. Simulation of water and solute transport in field soils with the LEACHP model. *Agric. Water Manage.* 44 225-245.
- Freijer, J I, Tiktak, A, Hassanizadeh, S M and Van der Linden, A M A. 1996. PESTRAS v. 3.1: A one-dimensional model for the assessment of the fate of pesticides in soil. Rep. No. 715501007. National Institute of Public Health and the Environment, Bilthoven, Netherlands, 130 pp.
- Guoy, V, Dur, J C, Clavet, R, Belamie, R and Chaplain, V. 1999. Influence of adsorption-desorption phenomena on pesticide run-off from soil using simulated rainfall. *Pesticide Sci.* 55(2) 175-182.
- Gustafson, D I. 1989. Groundwater Ubiquity Score: a simple method for assessing pesticide leachability. *Environmental Toxicology and Chemistry* 8 339-357.
- Hantush, M M, Marino, M A and Islam, M R. 2000. Models for leaching of pesticides in soils and groundwater. *J. Hydrol.* 227 66-83.
- Hutson, J L and Wagenet, R J. 1992. Leaching estimation and chemistry model v. 3.0. Dept. Soil, Crop and Atmospheric Sciences, Cornell University, New York.
- Huygen, J, Van Dam, J C and Kroes, J G. 2000. Modelling water flow and solute transport for agricultural and environmental management. SWAP graphical user interface, user manual. Alterra, Sub-department of Water Resources, Wageningen University and Research Centre.
- Inao, K and Kitamura, Y. 1999. Pesticide paddy field model (PADDY) for predicting pesticide concentrations in water and soil in paddy fields. *Pesticide Sci.* 55(1) 38-46.
- Jarvis, N. 1994. The MACRO Model (version 3.1). Technical Description and Sample Simulations. Swedish University of Agricultural Sciences, Department of Soil Science Reports and Dissertations, No. 19, Uppsala, Sweden, pp. 51.
- Jarvis, N, Brown, C and Granitz, E. 2000. Sources of error in model predictions of pesticide leaching: a case study using the MACRO model. *Agric Water Manage.* 44 247-262.
- Johanson, R C. 1983. New mathematical modeling system. In: R L Swann and A Eschenroeder (eds.) Fate of Chemicals in the Environment: Compartmental and Multimedia Models for Predictions. American Chemical Society, Washington, DC, USA, 125-147.
- Jury, W A and Masoud, G. 1989. Overview of organic chemical environmental fate and transport modeling approaches. In: Sawhney B and Brown K (eds.) Reactions and movement of organic chemicals in soils. SSSA No. 22, 271-301.

- Jury, W A, Spencer, W F and Farmer, W J. 1983. Behaviour assessment model for trace organics in soil. I. Model description. *J. Environ. Qual.* 12(4) 558-564.
- Jury, W A, Focht, D D and Farmer, W J. 1987. Evaluation of pesticide groundwater pollution potential from standard indices of soil chemical adsorption and biodegradation. *J. Environ. Qual.* 16 422-428.
- Klein, M, Hosant, J, Schafer, H, Ergraber, B and Ressler, H. 2000. Comparing and evaluating pesticide leaching models. Results of simulations with PELMO. *Agric. Water Manage.* 44 135-151.
- Leake, C R and Gatzweiler, E W. 1995. The relative importance of adsorption and desorption constants to the assessment of pesticide leaching potential in soil. In: *Pesticide movement to water, The British Crop Protection Council Monograph No. 62*, 19-24.
- Ma, Q L, Wauchope, R D, Hook, J E, Johnson, A W, Truman, C C, Dowler, C C, Gscho, G J, Davis, J G, Sumner, H R and Chandler, L D. 1998. GLEAMS, Opus and PRZM-2 model predicted versus measured runoff from a coastal plain loamy sand. *Trans. ASAE*, Jan/Feb 1998, 41(1) 77-88.
- McDonald, M G and Harbaugh, A W. 1988. A modular three-dimensional finite-difference groundwater flow model. *US Geological Survey Techniques of Water Resources Investigations*, Book 6, Chapter A1.
- Meeks, Y J and Dean, J D. 1990. Evaluating groundwater vulnerability to pesticides. *J. Water Resource Plng Mgmt* 116 693-707.
- Monteith, J L. 1977. Climate and efficiency of crop production in Britain. *Philos. Trans. R. Soc. London, Ser. B* 281 277-294.
- Nicholls, P H, Harris, G L and Brockie, D. 2000. Simulation of pesticide leaching at Vredepeel and Brimstone farm using the macropore model PLM. *Agric. Water Manage.* 44 307-315.
- Pang, X P and Letey, J. 1999. Pesticide leaching sensitivity to irrigation, salinity and N application: Model simulations. *Soil Sci.* 164(12) 922-929.
- Pepper, L, Debra, P and Bruseau, L. 1996. *Pollution science*. Academic Press, California.
- Rao, P S C, Hornsby, A G and Jessup, R E. 1985. Indices for ranking the potential for pesticide contamination of groundwater. *Proceedings of the Soil and Crop Science Society of Florida* 44 1-8.
- Rekolainen, S, Gouy, V, Francaviglia, R, Eklo, O M and Barlund, I. 2000. Simulation of soil water bromide and pesticide behaviour in soil with the GLEAMS model. *Agric. Water Manage.* 44 201-224.
- Rose, C and Ghadiri, H. 1992. An introduction to non-point source pollution modelling. In: Rose C and Ghadiri H (eds.) *Modelling chemical transport in soils - natural and applied contaminants*. Lewis Publishers, Florida, USA.
- Schlösser, S A and McCray, J E. 2002. Sensitivity of a pesticide leaching-potential index model to variations in hydrologic and pesticide-transport properties. *Environ. Geosci.* 9(2) 66-73.
- Scotter, D R, Thurbell, G W and Raats, P A C. 1967. Dispersion resulting from sinusoidal gas flow in porous materials. *Soil Sci.* 104 306-308.
- Smit, R, Leistra, M and Van den Berg, F. 1998. Estimation method for the volatilization of pesticides from plants. *Environmental Planning Bureau Series No. 4*, DLO Winand Staring Centre, Wageningen, The Netherlands.
- Trevisan, M, Errera, G, Goerlitz, G, Remy, B. and Sweeney, P. 2000a. PRZM-2 evaluation using Vredepeel data-set. *Agric. Water Manage.* 44 317-335.
- Trevisan, M, Errera, G, Vischetti, A and Walker, A. 2000b. Varleach evaluation. *Agric. Water Manage.* 44

Usda, S C S. 1972. National Engineering Handbook, Section 4: Hydrology. US Dept. of Agriculture, Soil Conservation Service, Washington, DC, pp. 548.

Van Clooster, M, Boesten, J J T I, Trevisan, M, Brown, C D, Capri, E, Eklo, O M, Gottesburen, B, Gouy, V and Van der Linder, A M A. 2000a. A European test of pesticide-leaching models: methodology and major recommendations. *Agric. Water Manage.* 44 1-19.

Van Clooster, M, Ducheyne, S, Dust, M and Vereecken, H. 2000b. Simulating pesticide dynamics with the WAVE-model. *Agric. Water Manage.* 44 371-388.

Walker, A. 1974. A simulation model for prediction of herbicide persistence. *J. Environ. Qual.* 3 396-401.

Williams, R J. 1998a. Modelling pesticide run-off to surface waters. I. Model theory and development. *Pesticide Sci.* 54(2) 113-120.

Williams, R J. 1998b. Modelling pesticide run-off to surface waters. I. Model application. *Pesticide Sci.* 54(2) 121-130.

Wischmeier, W H and Smith, D D. 1978. Predicting rainfall erosion losses. *Agriculture Handbook* 537, US Dept. of Agriculture, Washington, DC, pp. 58.

Woodrow, J E, Seiber, J N and Baker, L W. 1997. Correlation techniques for estimating pesticide volatilization flux and downwind concentrations. *Environmental Science and Technology* 31 523-529.

Yaron, B, Celvert, R and Prost, R. 1996. Soil pollution: Process and dynamics. Springer-Verlag, Berlin.

Zhou, J L, Rowland, S J, Mantoura, R F C and Harland, B J. 1995. Influence of the nature of particulate organic matter on the sorption of cypermethrin: implications on Koc correlations. *Environment International* 21(2) 187-195.

Surface Run-off

Adriaanse, P I, Beltman, W H J, Westain, E, Van Nierop, S. 1997. A proposed policy for differentiated hazard evaluation of pesticides in surface waters. DLO Winand Staring Centre, Report 141, Wageningen, Netherlands.

AEDG. 1992. Improving Aquatic Risk Assessment under FIFRA Report of the Aquatic Effects Dialogue Group. Resolve, Washington DC.

Baier, C, Hurle, K, Kirchhoff, J. 1985. Datensammlung zur Abschätzung des Gefährdungspotentials Von Pflanzenschutzmittelwirkstoffen für Gewässer. Parey Verlag, Hamburg.

Bascietto, J, Hinckley, D, Plafkin, J, Slimak, M. 1990. Ecotoxicity and ecological risk assessment. *Environ. Sci. Technol.* 24, 10-15.

Black, R W, Haggland, A L, Voss, F D. 2000. Predicting the probability of detecting organochlorine pesticides and polychlorinated biphenyls in stream systems on the basis of land use in the Pacific Northwest, USA. *Environ. Toxicol. Chem.* 19, 1044-1054.

Blanchard, P E, Lerch, R N. 2000. Watershed vulnerability to losses of agricultural chemicals: interactions of chemistry, hydrology and land-use. *Eviron. Sci. Technol.* 34, 3315-3322.

Capel, P D, Larson, S J. 2001. Effect of scale on the behaviour of atrazine in surface waters. *Environ. Sci. Technol.* 35, 648-656.

Castelle, A J, Johnson, A W, Conolly, C. 1994. Wetland and Stream Buffer Size Requirements-A Review. *J Environ Qual* 23:878-882.

Cole, J T, Baird, J H, Basta, N T, Huhnke, R L, Storm, D E, Johnson, G V, Payton, M E, Smolen, M D, Martin, D L, Cole, J C. 1997. Influence of buffers on pesticide and nutrient runoff from Bermudagrass turf. *J. Environ. Qual.* 26, 1589-1598.

Cooper, C M. 1993. Biological effects of agriculturally derived surface-water pollutants on aquatic systems - a review. *J. Environ. Qual.* 22, 402-408.

Dabrowski, J M, Peall, S K C, Reinecke, A J, Liess, M, Schulz, R. 2001. Runoff-related pesticide input into the Lourens River, South Africa: Basic data for exposure assessment and risk mitigation at the catchment scale. *Wat. Air Soil Pollut.* 135, 265-283

Dabrowski, J M, Peall, S K C, Van Niekerk, A, Reinecke, A J, Day, J A, Schulz, R. 2002. Predicting runoff-induced pesticide input in agricultural sub-catchment surface waters: linking catchment variables and contamination. *Water Research.* 36, 4975-4984

Dabrowski, J M, Schulz, R. (2003). Predicted and measured levels of azinphosmethyl in the Lourens River, South Africa: comparison of runoff and spray drift. *Environ. Toxicol. Chem.* 22(3), 494-500

Dallas, H F, Day, J A. 1993. The Effect of Water Quality Variables on Riverine Ecosystems: A Review. Water Research Commission, Pretoria

Davies, H E, Peall, S K C. 1997. A study of pesticide residue levels in farm dams in the Western Cape, South Africa. *Fitz Patrick Institute* 1-12.

Domagalski, J L, Dubrovsky, N M, Kratzer, C R. 1997. Pesticides in the San Joaquin River, California: Inputs from the dormant sprayed orchards. *J. Environ. Qual.* 26, 454-465.

Donigan JR. A S, Huber, W C. 1991. Modeling of non-point source water quality in urban and non-urban areas. United States Environmental Protection Agency, Environmental Research Laboratory, Office of Research and Development. Athens, Georgia. EPA /600/3 -91/039.

Erstfeld, K M. 1999. Environmental fate of synthetic pyrethroids during spray drift and field runoff treatments in aquatic microcosms. *Chemosphere* 39, 1737-1769.

Everts, J W. 1997. Ecotoxicology for risk assessment in arid zones: Some key issues. *Arch. Environ. Contam. Toxicol.* 32, 1-10.

Felsot, A S, Mitchell, J K, Kenimer, A L. 1990. Assessment of management practices for reducing pesticide runoff from sloping cropland in Illinois. *J. Environ. Qual.* 19: 539-545

FOCUS. 1997. Surface Water Models and EU Registration of Plant Protection Products. European Commission Document 6476/VI/96

FOCUS. 2001. *FOCUS Surface Water Scenarios in the EU Evaluation Process under 91/414/EEC*. Report of the FOCUS Working Group on Surface Water Scenarios, EC Document Reference SANCO/4802/2001-rev.1. 221 pp.

Flury, M. 1996. Experimental Evidence of Transport of Pesticides through Field Soils - A Review. *J. Environ. Qual.* 25, 25-45.

Ganzelmeier, H, Rautmann, D, Spangenberg, R, Streloke, M, Herrmann, M, WenZelburger, H J, Walter, H F. 1995. Studies of the spray drift of plant protection products. *Mitteilungen aus der Biologischen Bundesanstalt für Land- und Forstwirtschaft*, Blackwell Scientific Publisher, Berlin.

Gilbert, A J, Bell, G J. 1988. Evaluation of the drift hazards arising from pesticide spray application. *Aspects Appl. Biol.* 17, 363-376.

Greichus, Y A, Greichus, A, Amman, B D, Call, D J, Hamman, D C D, Pott, R M. 1977. Insecticides polychlorinated biphenyls and metals in african lake ecosystems. I. Hartbeespoort Dam, Transvaal and Voelvlei Dam, Cape Province, Republic of South Africa. *Arch. Environ. Contam. Toxicol.* 6, 371-383.

Grobler, D F. 1994. A note on PCBs and chlorinated hydrocarbon pesticide residues in water, fish and sediment from the Olifants River, Eastern Transvaal, South Africa. *Water SA* 20, 187-194.

- Groenendijk, P., Van der Kolk, J. W. H., Travis, K. Z. 1994. Prediction of exposure concentration in surface waters. In *Freshwater field tests for hazard assessment of chemicals*, eds. I. R. Hill, F. Heimbach, P. Leeuwangh and P. Matthiessen, pp. 105-125. Lewis Publisher, Boca Raton.
- Grunwald, S., Norton, L. D. 2000. Calibration and validation of a non-point source pollution model. *Agric. Wat. Manage.* 45, 17-39.
- Hand, L. H., Kuet, S. F., Lane, M. C. G., Maund, S. J., Warinton, J. S., Hill, I. R. 2001. Influences of aquatic plants on the fate of the pyrethroid insecticide lambda-cyhalothrin in aquatic environments. *Environ. Toxicol. Chem.* 20 (8), 1740-1745.
- Hassett, A. J., Viljoen, P. T., Liebenberg, J. J. E. 1987. An assessment of chlorinated pesticides in the major surface water resources of the Orange Free State during the period september 1984 to september 1985. *Water SA* 13, 133-136.
- Kreuger, J. 1998. Pesticides in stream water within an agricultural catchment in southern Sweden, 1990-1996. *Sci. Total Environ.* 216, 227-251.
- Jones, R. L., Mangels, G. 2002. Review of the validation of models used in Federal Insecticide, Fungicide and Rodenticide Act environmental exposure assessments. *Environ. Toxicol. Chem.* 21(8) 1535-1544.
- Line, D. E., Osmond, D. L., Coffey, S. W., McLaughlin, R. A., Jennings, G. D., Gale, J. A. 1997. Non-point sources. *Wat. Environ. Res.* 69, 844-860.
- Loague, K., Corwin, D. L., Ellsworth, T. R. 1998. The challenge of predicting non-point source pollution. *Environ. Sci. Technol.* 32, 130-133.
- Longley, M., Sotherton, N. W. 1997. Measurements of pesticide spray drift deposition into field boundaries and hedgerows: 2. Autumn applications. *Environ. Toxicol. Chem.* 16, 173-178.
- Lutz, W. 1984. Calculation of stormwater discharge using catchment variables (in German). Communications, Institute of Hydrology and Water Research, University of Karlsruhe Vol. 24.
- Maniak, U. 1992. Regionalization of parameters for stormwater flow curves. In: *Regionalization in Hydrology* (in German). DFG, Communications of the Commission for Water. 11, 325-332.
- Merkle, M. G., Bovey, R. W. 1974. *Movement of pesticides in surface water*, in: W. D. Guenzi (ed.), *Pesticides in soil and water*, Madison, Wisconsin, pp. 99-105.
- Mian, L. S., Mulla, M. S. 1992. Effects of Pyrethroid Insecticides on Nontarget Invertebrates in Aquatic Ecosystems. *J. Agric. Ent.* 9, 73-98.
- Moore, M. T., Rodgers Jr, J. H., Smith Jr., S., Cooper, C. M. 2001a. Mitigation of metolachlor-associated agricultural runoff using constructed wetlands in Mississippi, USA. *Agric. Ecosys. Environ.* 84, 169-176.
- Moore, M. T., Bennett, E. R., Cooper, C. M., Smith Jr, S., Shields Jr, F. D., Milam, C. D., Farris, J. L. 2001b. Transport and fate of atrazine and lambda-cyhalothrin in an agricultural drainage ditch in the Mississippi Delta, USA. *Agric. Ecosys. Environ.* 87, 309-314.
- Patty, L., Real, B., Gril, J. J. 1997. The use of grassed buffer strips to remove pesticides, nitrate and soluble phosphorus compounds from runoff water. *Pestic. Sci.* 49, 243-251.
- Reus, J., Lennerste, C., Bockstaller, C., Fomsgaard, I., Gutsche, V., Lewis, K., Nilsson, C., Pussemier, L., Trevisan, M., Van der Werf, H., Alfaro, F., Blumel, S., Isart, J., McGrath, D., Seppälä, T. 1999. Comparing Environmental Risk Indicators for Pesticides. Results of the European CAPER Project. Centre for Agriculture and Environment Utrecht, CLM 426, 184 pp.

- Roux, D J, Badenhorst, J E, Du Preez, H H, Steyn, G J. 1994. Note on the occurrence of selected trace metals and organic compounds in water, sediment and biota of the Crocodile River, Eastern Transvaal, South Africa. *Water SA* 20, 333-340.
- Schulz, R. 2001a. Comparison of spraydrift- and runoff-related input of azinphos-methyl and endosulfan from fruit orchards into the Lourens River, South Africa. *Chemosphere* 45, 543-551.
- Schulz, R. 2001b. Rainfall-induced sediment and pesticide input from orchards into the Lourens River, Western Cape, South Africa: importance of a single event. *Water Res.* 35, 1869-1876.
- Schulz, R, Liess, M. 2001. Toxicity of aqueous-phase and suspended particle-associated fenvalerate: chronic effects following pulse-dosed exposure of *Limnephilus lunatus* (Trichoptera). *Environ. Toxicol. Chem.* 20, 185-190.
- Schulz, R, Peall, S K C. 2001. Effectiveness of a constructed wetland for retention of non-point-source pesticide pollution in the Lourens River catchment, South Africa. *Environ. Sci. Technol.* 35, 422-426.
- Schulz, R., Peall, S K C, Dabrowski, J M, Reinecke, A J. 2001. Current-use insecticides, phosphates and suspended solids in the Lourens River, Western Cape, during the first rainfall event of the wet season. *Water SA* 27, 65-70.
- Schulz, R, Hahn, C, Bennett, E R, Dabrowski, J M, Thiere, G, Peall, S K C. 2003. Fate and effects of azinphos-methyl in a flow-through wetland in South Africa. *Environ. Sci. Technol.* 37, 2139-2144.
- Singh, P, Jones, R L. 2002. Comparison of Pesticide Root Zone Model 3.12: Runoff predictions with field data. *Environ. Toxicol. Chem.* 21(8) 1545-1551.
- Solomon, K R, Baker, D B, Richards, R P, Dixon, K R, Klaine, S J, La-Point, T W, Kendall, R J, Weisskopf, C P, Giddings, J M, Giesy, J P, Hall, L W J, Williams, W M. 1996. Ecological risk assessment of atrazine in North American surface waters. *Environ. Toxicol. Chem.* 15, 31-76.
- Squillace, P J, Caldwell, J P, Schulmeyer, P M, Harvey, C A. 1996. Movement of Agricultural Chemicals Between Surface Water and Ground Water, Lower Cedar River Basin, Iowa. United States Government Printing Office, Washington. Environmental Protection Agency, Washington.
- Trevisan, M, Errera, G, Goerlitz, G, Remy, B, Sweeney, P. 2000. Modelling ethoprophos and bentazone fate in a sandy humic soil with primary pesticide fate model PRZM-2. *Agric. Wat. Man.* 44, 317-335.
- Verro, R, Calliera, M, Maffioli, G, Auteri, D, Sala, S, Finizio, A, Vighi, M. 2002. GIS-based system for risk assessment of agricultural chemicals. 1. Methodological approach. *Environ. Sci. Technol.* 36(7) 1532-1538.
- Walklate, P J. 1992. A simulation study of pesticide drift from an air-assisted orchard sprayer. *J. Agric. Eng. Res.* 51, 263-283.
- Wauchope, R D. 1996. Pesticides in Runoff: Measurement, modelling and mitigation. *J. Environ. Sci. Health. Part B* 31, 337-344.
- Wilcox, B P, Wood, M K. 1989. Factors influencing interrill erosion from semiarid slopes in New Mexico (USA). *J. Range Manage.* 42, 66-70.
- Williams, R J, Brooke, D, Matthiesen, P, Mills, M, Turnbull, A, Harrison, R M. 1995. Pesticide transport to surface waters within an agricultural catchment. *J. Instit. Wat. Environ. Manage.* 9, 72-81.
- Zullei-Seibert, N, Skark, C. 1993. Verhalten von Pflanzenschutzmitteln auf dem Transportpfad "Oberflächenwasser-Uferfiltrat-Grundwasser". *Berichte aus der Dortmunder Energie und Wasserversorgung* 376.

Soil processes

Agricultural Chemical Usage: Field crops summary (1990). National Agricultural Statistics Service, Economic Research Service, U.S. Dept. of Agric., Washington DC, pp 1-154.

Aderhold, D and Nordmeyer, H (1995). Leaching of herbicides in soil macropores as a possible reason for groundwater contamination. BCPC Mono No. 62, pp 217-222.

Allen, R and Walker, A (1983). Influence of soil characteristics on herbicide degradation rates. Report: National Vegetable Res. Stn., England, 1982, 128 pp.

Anderson, W P (1983). Weed science: Principles (2nd edn). West Publishing Co., New York.

Anderson, J R; Stephenson, G R and Corke, C T (1980). Atrazine and cyanazine activity in Ontario and Manitoba soils. Can. J. Soil Sci., Vol. 60, pp 773-781.

Appleby, A P (1985). Factors in examining fate of herbicides in soil with bioassays. Weed Sci., Vol. 33(suppl. 2), pp 2-6.

Armstrong, D E; Chesters, G and Harris, R F (1967). Atrazine hydrolysis in soil. Soil Sci. Soc. Am. Proc., Vol. 31, pp 61-66.

Armstrong, D E and Chesters, G (1968). Adsorption catalyzed chemical hydrolysis of atrazine. Environ. Sci. Technol., Vol. 2, pp 683-689.

Bailey, G W and White, J L (1970). Factors influencing the adsorption, desorption, and movement of pesticides in soil. In: F.A. Gunther (ed). The triazine herbicides. Residue Reviews, Vol. 32, pp 29-92.

Best, J A and Weber, J B (1974). Disappearance of s-triazines as affected by soil pH using a balance sheet approach. Weed Sci., Vol. 22, pp 364-373.

Boesten, J J T I and Van der Linden, A M A (1991). Modelling the influence of sorption and transformation on pesticide leaching and persistence. J. Environ. Qual., Vol. 20, pp 425-435.

Bohn, H L; McNeal, B L and O'Connor, G A (1985). Soil chemistry. John Wiley & Sons Inc., New York.

Bollag, J-M and Liu, S-Y (1990). Biological transformation processes of pesticides. In: H H Cheng (ed). Pesticides in the soil environment: Processes, impacts and modelling. Book no. 2 in Soil Science Society of America Book Series. SSSA Inc., Madison, Wisconsin, USA.

Brady, N C (1974). The nature and properties of soils (8th edn). Collier-McMillan, London.

Briggs, G G (1981). Theoretical and experimental relationships between soil adsorption, octanol-water partition coefficients, water solubilities, bioconcentration factors and the parachor. J. Agric. Food Chem., Vol. 29, pp 1050-1059.

Briggs, G G (1983). Factors affecting pesticide degradation in soil. In: Pesticide residues. Ministry of Agriculture Fisheries and Food Ref. Book No. 347. HMSO, London.

Briggs, G G (1984). Factors affecting the uptake of soil-applied chemicals by plants and other organisms. BCPC Mono. No. 27, pp 35-47.

Buhler, D D; Randall, G W; Koskinen, W C and Wyse, D L (1993). Atrazine and alachlor losses from subsurface tile drainage of a clay loam soil. J. Environ. Qual., Vol. 22, pp 583-588.

Burnside, O C; Fenster, C R and Wicks, G A (1963). Dissipation and leaching of monuron, simazine and atrazine in Nebraska soils. Weeds, Vol. 11, pp 209-213.

Cain, R B and Head, I M (1991). Enhanced degradation of pesticides: Its biochemical and molecular biological basis. In: A Walker (ed). Pesticides in soil and water: Current Perspectives. BCPC Mono. No. 47, pp 23-40.

- Carsel, R F; Mulkey, L A; Lorber, M N and Baskin, L B (1985). The Pesticide Root Zone Model (PRZM): a procedure for evaluating pesticide leaching threats to groundwater. *Ecol. Model.*, Vol. 30, 150 pp.
- Caverley, D J (1983). Significance of residues to subsequent crops. *Pesticide Residues*, Ministry of Agriculture Fisheries and Food Reference Book No. 347, pp 112-121. HMSO, London.
- Coetzee, J P and Vahrmeijer, J T (undated). The adsorption of atrazine by different clay minerals and its influence on the mobility of atrazine. University of Potchefstroom, Potchefstroom, South Africa.
- Cohen, S Z; Eoden, C and Lorber, M N (1986). Monitoring groundwater for pesticides. p 170-196. In: Y W Garner et al. (eds.). *Evaluation of pesticides in groundwater*. Am. Chem. Soc. Symp. Ser. 315. ACS, West End Station, Washington D.C.
- Colbert, F O; Volk, V V and Appleby, A P (1975). Sorption of atrazine, terbutryn and GS-14254 on natural and lime-amended soils. *Weed Sci.*, Vol. 23, pp 390-394.
- Dao, T H and Lavy, T L (1978). Atrazine adsorption on soil as influenced by temperature, moisture content and electrolyte concentration. *Weed Sci.*, Vol. 26, pp 303-308.
- Duffy, M J (1991). The characterization of herbicide persistence. BCPC Mono. No. 47, pp 85-92.
- Ellis, J F (1992). Herbicide development and marketing of weed control in the United States of America. *Proc. First Internat. Weed Control Conf.*, Vol. 1, pp 74-82.
- Eagle, D J (1976). Soil texture classification for the adjustment of herbicide dose. *Proc. BCPC Conf.: Weeds*, pp 981-988.
- Eagle, D J (1978). Interpretation of soil analysis for herbicide residues. *Proc. 1978 BCPC Conf.: Weeds*, Vol. 2, pp 535-539.
- Eagle, D J (1983a). Matching herbicide dose to soil type. *Pesticide Residues*, Ministry of Agriculture Fisheries and Food Reference Book No. 347, pp 86-93. HMSO, London.
- Eagle, D J (1983b). An agronomic view of environmental effects on the performance of soil-applied herbicides. *Aspects Appl. Biol.*, Vol. 4, pp 389-394.
- Ehlers, J G; Reinhardt, C F and Nel, P C (1987). Atrazine activity under field conditions. *Appl. Plant Sci.*, Vol. 1, pp 57-60.
- Ehlers, J G; Reinhardt, C F and Nel, P C (1988). Effect of certain soil factors on the activity of atrazine. *S. Afr. J. Plant Soil*, Vol. 5, pp 32-36.
- Enfield, C G and Yates, S R (1990). Organic chemical transport to groundwater. In: C C Cheng (ed). *Pesticides in the soil environment: Processes, impacts and modelling*. Book no. 2 in Soil Science Society of America Book Series. SSSA Inc., Madison, Wisconsin, USA.
- Erickson, L E and Lee, K H (1989). Degradation of atrazine and related s-triazines. *Crit. Rev. Environ. Control*, Vol. 19, pp 1-14.
- Esser, H O; Dupuis, G; Ebert, E; Marco, G J and Vogel, C (1975). S-triazines. In: P C Kearney and D D Kaufman (eds). *Herbicides: Chemistry, degradation and mode of action*. Vol 1 (2nd edn). Marcel-Dekker, Inc., New York.
- Fleming, Gwen F; Wax, L M and Simmons, F W (1992). Leachability and efficacy of starch-encapsulated atrazine. *Weed Technol.*, Vol. 6, pp 297-302.
- Fouché, P S and Brandt, M P (1973). Die aard van anorganiese amorfe materiaal in 'n seskwioksiedgrond in Transvaal. *Proc. Electron Microscopy Soc. of S. Afr.*, Vol. 3, pp 105-106.

- Frissel, M J (1961). The adsorption of some organic compounds, especially herbicides on clay minerals. *Verslag Landbouwk. Onderzoek*, Vol. 76, pp 3-6.
- Gamble, D S and Khan, S U (1985). Atrazine hydrolysis in soils, catalysis by the acidic functional groups of fulvic acid. *Can. J. Soil Sci.*, Vol. 65, pp 435-443.
- Gottesbüren, B, Pestemer, W, Wang, K, Wischniewsky, M B and Zhao, J (1991). Concept, structure and validation of the expert system HERBASYS (herbicide advisory system) for selection of herbicides, prognosis of persistence and effects on succeeding crops. In: A Walker (ed). *Pesticides in soils and water: Current perspectives*. BCPC Mono. No. 47, pp 129-138.
- Green, R E and Obien, S R (1969). Herbicide equilibrium in soils in relation to soilwater content. *Weed Sci.*, Vol. 17, pp 514-519.
- Grover, R (1965). Influence of organic matter, texture and available water in the toxicity of simazine in soil. *Weeds*, Vol. 15, pp 148-151.
- Hall, J K, Murray, M R and Hartwig, N L (1989). Herbicide leaching and distribution in tilled and untilled soil. *J. Environ. Qual.*, Vol. 18, pp 439-445.
- Hance, R J (1979). Influence of adsorption on the decomposition of pesticides. *Soc. of Chem. Ind., London, Mono. No. 37*, pp 92-104.
- Hance, R J (1983). Processes in soil which control the availability of herbicides. *Proc. 10 th Internat. Congress of Plant Protection*, pp 537-544.
- Harris, G R and Hurle, K (1979). The effect of plant-induced changes of pH upon the adsorption and phytotoxicity of s-triazine herbicides. *Weed Res.*, Vol. 19, pp 343-349.
- Harris, C I, Woolson, E A and Hummer, B E (1969). Dissipation of herbicides at three soil depths. *Weed Sci.*, Vol. 17, pp 27-31.
- Harrison, G W, Weber, J B and Baird, J V (1976). Herbicide phytotoxicity as affected by selected properties of North Carolina soils. *Weed Sci.*, Vol. 24, pp 120-125.
- Hartley, G S (1976). Physical behaviour in the soil. In: L J Audus (ed). *Herbicides: Physiology, Biochemistry, Ecology* (2 edn). Academic Press, London.
- Hayes, M H B (1970). Adsorption of triazine herbicides on soil organic matter, including a short review on soil organic matter chemistry. In: F A Gunther, *The triazine herbicides*. *Residue Rev.*, Vol. 32, pp 131-174.
- Hollis, J M (1991). Mapping the vulnerability of aquifers and surface waters to pesticide contamination at the national/regional scale. In: A Walker (ed). *Pesticides in soils and water*. BCPC Mono. No. 47, pp 165-174.
- Hollis, J M, Keay, C A, Hallett, S H and Gibbons, J W (1995). Using CATCHIS to assess the risk to water resources from diffusely applied pesticides. BCPC Mono. No. 62, pp 345-350.
- Hubbs, C W and Lavy, T L (1990). Dissipation of norflurazon and other persistent herbicides in soil. *Weed Sci.*, Vol. 38, pp 81-88.
- Hugo, K J (1994). Leaching of atrazine and terbuthylazine in some South African soils. M Sc (Agric) thesis. University of Pretoria, Pretoria, South Africa.
- Hurle, K B and Freed, V H (1972). Effect of electrolytes on the solubility of some 1,3,5-triazines and substituted ureas and their adsorption on soil. *Weed Res.*, Vol. 12, pp 1-10.
- Jarvis, T D (1995). Simulation modelling and UK regulatory decision-making: Perfect partners or warring factions? BCPC Mono. No. 62, pp 357-362.

- Kamffer, F H (1997). (Incomplete Ph D thesis). Dept. of Botany, University of Pretoria, Pretoria, South Africa.
- Kaufman, D D and Kearney, P C (1970). Microbial degradation of s-triazine herbicides. *Residue Reviews*, Vol. 32, pp 235-265.
- Koskinen, W C and Harper, S S (1990). The retention process: Mechanisms. In: C C Cheng (ed). *Pesticides in the soil environment: Processes, impacts and modelling*. Book no. 2 in Soil Science Society of America Book Series. SSSA Inc., Madison, Wisconsin, USA.
- Kearney, P C and Kaufman, D D (1988). *Herbicides: Chemistry, degradation, and mode of action*. Vol 3. Marcel Dekker, Inc. New York.
- Kononova, M M (1966). *Soil organic matter*. Pergamon Press, Oxford, England.
- Krueger, R F and Seiber, J N (1984). *Treatment and disposal of pesticide wastes*. American Chemical Society, Washington DC.
- Laird, D A; Yen, P Y; Koskinen, W C and Dowdy, R H (1994). Sorption of atrazine on soil clay components. *Environ. Sci. Technol.*, Vol. 28, pp 1054-1061.
- Lavy, T L (1968). Micromovement mechanisms of s-triazines in soil. *Soil Sci. Soc. Am. Proc.*, Vol. 32, pp 377-380.
- Lavy, T L; Mattice, J D; Massey, J H; Skulman, B W; Senseman, S A; Gbur, E E and Barrett, M R (1996). Long-term in situ leaching and degradation of six herbicides aged in subsoils. *J. Environ. Qual.*, Vol. 25, pp 1268-1279.
- Lavy, T L; Roeth, F W and Fenster, C R (1973). Degradation of 2,4-D and atrazine at three soil depths in the field. *J. Environ. Qual.*, Vol. 1, pp 132-137.
- Leake, C R and Gatzweiler, E W (1995). The relative importance of adsorption and desorption constants to the assessment of pesticide leaching potential in soil. *BCPC Mono No. 62*, pp 19-24.
- Leeds-Harrison, P B (1995). The movement of water and solutes to surface and ground waters. *BCPC Mono No. 62*, pp 3-12.
- Leonard, R A (1990). Movement of pesticides into surface waters. In: H H Cheng (ed). *Pesticides in the soil environment: Processes, impacts and modelling*. Book no. 2 in Soil Science Society of America Book Series. SSSA Inc., Madison, Wisconsin, USA.
- Leonard, R A; Shirmohammadi, A; Johnson, A W and Marti, L R (1988). Pesticide transport in shallow groundwater. *Transactions of the ASAE*, Vol. 31, pp 776-788.
- Madhun, Y A and Freed, V H (1990). Impact of pesticides on the environment. In: C C Cheng (ed). *Pesticides in the soil environment: Processes, impacts and modelling*. Book no. 2 in Soil Science Society of America Book Series. SSSA Inc., Madison, Wisconsin, USA.
- Marshall, R J; Nel, P C and Smit, N S H (1982). Atrazine phytotoxicity to sorghum as affected by soil factors and temperature. *Crop Prod.*, Vol. 11, pp 147-149.
- McGlamery, M D and Slife, F W (1965). The adsorption and desorption of atrazine as affected by pH, temperature, and concentration. *Weeds*, Vol. 14, pp 237-239.
- Meinhardt, H.R., 2003. Evaluation of Predictive Models for Pesticide behaviour in South African Soils, WRC Report no. 999/1/03, ISBN 1868459683, WRC, Pretoria, South Africa, pp 90.
- Moorman, T B and Harper, S S (1989). Transformation and mineralization of metribuzin in surface and subsurface horizons of a Mississippi Delta soil. *J. Environ. Qual.*, Vol. 18, pp 302-306.
- Nel, P C (1975). The role of soil type and the degree of leaching in atrazine efficacy. *Crop Prod.*, Vol. 4, pp 83-86.

Nel, P C and Reinhardt, C F (1984). Factors affecting the activity of atrazine in plants and soil. *S. Afr. J. Plant Soil*, Vol. 1, pp 67-72.

Nicholls, P H; Briggs, G G and Evans, A A (1984). The influence of water solubility on the movement and degradation of simazine in a fallow soil. *Weed Res.*, Vol. 24, pp 37-49.

Nicholls, P H and Hall, D G M (1995). Use of the Pesticide Leaching model (PLM) to simulate pesticide movement through macroporous soils. *BCPC Mono.* No. 62, pp 187-192.

Nicholls, P H; Walker, A and Baker, R J (1982). Measurement and simulation of the movement and degradation of atrazine and metribuzin in a fallow soil. *Pestic. Sci.*, Vol. 12, pp 484-494.

Osteen, C D and Symeda, P I (1989). Agricultural pesticide use trends and policy issues. *Agricultural Economic Report 622*, Economic Research Service, U.S. Dept. of Agric., Washington D.C.

Pestemer, W and Günther, P (1995). Use of a pesticide monitoring system (PEMOSYS) for the risk assessment of pesticide leaching potential - concept for post-registration activities. *BCPC Mono.* No. 62, pp 351-356.

Reinhardt, C F and Hugo, K J (1997). Simple regression models for the qualitative prediction of atrazine and terbuthylazine leaching. *S Afr. J. Plant Soil*, Vol. 14(2), in press.

Reinhardt, C F and Nel, P C (1989). Importance of selected soil properties on the bioactivity of alachlor and metolachlor. *S. Afr. J. Plant Soil*, Vol. 6, pp 120-123.

Riley, D (1995). Assessment, acceptability and regulation of risk from residues in water. *BCPC Mono.* No. 62, pp 335-343.

Riley, D and Morrod, R S (1976). Relative importance of factors influencing the activity of herbicides in soil. *BCPC Conf.: Weeds*, pp 971-980.

Russo, E; Fava, A; Achilli, F and Sassi, E (1993). Groundwater contamination by pesticides: extensive field studies. *Proc. IX Symposium Pesticide Chemistry: Mobility and degradation of xenobiotics*. Piacenza, Italy, pp 707-718.

Schiavon, M (1988). Studies of the movement and the formation of bound residues of atrazine, of its chlorinated derivatives, and of hydroxyatrazine in soil using ¹⁴C ring-labeled compounds under outdoor conditions. *Ecotoxicology and Environmental Safety*, Vol. 15, pp 55-61.

Seiler, A and Mühlebach, F (1995). Good farming practices to reduce residues of atrazine in ground and surface water. *BCPC Mono.* No. 62, pp 389-396.

Senseman, S A; Lavy, T L; Mattice, J D and Daniel, T C (1996). Monitoring of groundwater for pesticides at selected mixing/loading sites in Arkansas. *Environ. Sci. Technol.* (in press).

Severn, D J and Ballard, G (1990). Risk/Benefit and regulations. In: C C Cheng (ed). *Pesticides in the soil environment: Processes, impacts and modelling*. Book no. 2 in *Soil Science Society of America Book Series*. SSSA Inc., Madison, Wisconsin, USA.

Smit, N S H and Nel, P C (1977). The activity of atrazine on two South African soils. *Crop Prod.*, Vol. 6, pp 67-71.

Smit, N S H; Nel, P C and Fölscher, W J (1979). Effect of pH on availability, residual activity and desorption of atrazine in soils with the same clay content. *Crop Prod.*, Vol 8, pp 125-129.

Smit, N S H; Nel, P C and Fölscher, W J (1980). Factors affecting the sorption and degradation of atrazine in soil. *Crop Prod.*, Vol 9, pp 135-139.

Southwick, L M; Willis, G H; Johnson, D C and Selim, H M (1995). Leaching of nitrate, atrazine and metribuzin from sugar cane in southern Louisiana. *J. Environ. Qual.*, Vol. 24, pp 684-690.

Spencer, W F and Clith, M M (1973). Pesticide volatilization as related to water loss from soil. *J. Environ. Qual.*, Vol. 2, pp 284-288.

Swain, D J (1981). Atrazine dissipation in irrigated sorghum cropping in southern New South Wales. *Weed Res.*, Vol. 21, pp 13-21.

Talbert, R E and Fletchall, O H (1965). The adsorption of some s-triazines in soils. *Weeds*, Vol. 13, pp 46-52.

Tooby, T E and Marsden, P K (1991). Interpretation of environmental fate and behaviour data for regulatory purposes. *BCPC Mono. No. 47*, pp 3-10.

Van Biljon, J J (1991). Factors affecting the selectivity of metolachlor in maize *Zea mays* L. Ph D thesis, University of Pretoria, Pretoria, South Africa.

Van Genuchten, M Th; Ralston, D E and Germann, P F (1990). Transport of water and solutes in macropores. *Geoderma*, Vol. 46, special issue.

Wagenet, R J and Hutson, J L (1992). *LEACHM* Leaching estimation and chemistry model. Continuum Vol. 2, Water Resources Institute, Cornell University, Ithaca, NY, 148 pp.

Wagenet, R J and Rao, P S C (1990). Modelling pesticide fate in soil. In: H H Cheng (ed). *Pesticides in the soil environment: Processes, impacts and modelling*. Book no. 2 in *Soil Science of America Book Series*. SSSA Inc., Madison, Wisconsin, USA.

Walker, A; Welch, S J and Turner, I J (1995). Studies of time-dependent sorption processes in soils. *BCPC Mono No. 62*, pp 13-18.

Walker, A (1980). Activity and selectivity in the field. In: R J Hance (ed). *Interactions between herbicides and the soil*. Academic Press, London.

Walker, A (1987). Herbicide persistence in soil. *Rev. Weed Sci.*, Vol. 3, pp 1-17.

Walker, A and Allen, R (1984). Influence of soil and environmental factors on pesticide persistence. In: R J Hance (ed). *Soils and crop protection chemicals*. *BCPC Mono. No. 27*, pp 89-100.

Walker, A and Zimdahl, R L (1981). Simulation of the persistence of atrazine, linuron and metolachlor in soil at different sites in the U.S.A. *Weed Res.*, Vol. 21, pp 255-265.

Walker, A (1989). Factors influencing the variability in pesticide persistence in soils. *Aspects of Applied Biology*, Vol. 12, pp 159-172.

Weber, J B (1970a). Mechanisms of adsorption of s-triazines by clay colloids and factors affecting plant availability. In: F.A. Gunther (ed). *The triazine herbicides*. *Residue Reviews*, Vol. 32, pp 93-130.

Weber, J B (1970b). Adsorption of s-triazines by montmorillonite as a function of pH and molecular structure. *Soil Sci. Soc. Am. Proc.*, Vol. 34, pp 401-404.

Weber, J B (1972). Interaction of organinic pesticides with particulate matter in aquatic and soil systems. In: R F Gould (ed). *Fate of organic pesticides in the aquatic environment*. American Chemical Society, Washington, D.C.

Weber, J B (1991a). Fate and behaviour of herbicides in soil. *Appl. Plant Sci.*, Vol. 5, pp 28-41.

Weber, J B (1991b). Potential for ground water contamination from selected herbicides: a herbicide/soil ranking system. *Proc. Southern Weed Sci. Soc.*, Vol. 44, pp 45-57.

Weber, J B; Shea, P H and Weed, S B (1986). Fluridone retention and release in soils. *Soil Sci. Soc. Am. J.*, Vol 50, pp 582-588.

Weber, J B; Weed, S B and Ward, T M (1969). Adsorption of s-triazines by soil organic matter. *Weed Sci.*, Vol. 17, pp 417-421.

Weber, J B and Whitacre, D M (1982). Mobility of herbicides in soil columns under saturated- and unsaturated-flow conditions. *Weed Sci.*, Vol. 30, pp 579-584.

Wolfe, N L; Mingelgrin, U and Miller, G C (1990). Abiotic transformations in water, sediments and soil. In: H H Cheng (ed). *Pesticides in the soil environment: Processes, impacts and modelling*. Book no. 2 in Soil Science Society of America Book Series. SSSA Inc., Madison, Wisconsin, USA.

Wood, M; Harold, J; Johnson, A and Hance, R J (1991). The potential for atrazine degradation in aquifer sediments. *BCPC Mono. No. 47*, pp 175-182.

Yamane, V K and Green, R E (1972). Adsorption of ametryne and atrazine on an oxisol, montmorillonite, and charcoal in relation to pH and solubility effects. *Soil Sci. Soc. Am. Proc.*, Vol. 36, pp 58-64.

Baer-U; Calvet-R; 1996. Simulation of the persistence of atrazine and sulcotrione in two cultivated soils. *Association Nationale pour la Protection des Plantes (ANPP)*; Paris, France

Bailey, GW and White JL, (1970), *Residue reviews*, 32, 29-92

Barbash, JE and Reseck, EA, (1996). *Pesticides in Ground Water*, Ann Arbor Press: Chelsea, MI pp 588.

Borchers-U; Peters-B; Overath-H; Walker-A (ed.); Allen-R (ed.); Bailey-SW (ed.); Blair-AM (ed.); Brown-CD (ed.); Gunther-P (ed.); Leake-CR (ed.); Nicholls-PH; 1995. Limits and possibilities of using computer models to predict fate and behaviour of pesticides in soil. Pesticide movement to water. Proceedings of a symposium held at the University of Warwick, Coventry, UK, on 3-5 April 1995. 1995, 281-286; *BCPC Monograph No. 62*; 7 ref.. Publisher information: British Crop Protection Council (BCPC); Farnham; UK

Brown-CD; Baer-U; Gunther-P; Trevisan-M; Walker-A; 1996. Ring test with the models LEACHP, PRZM-2 and VARLEACH: variability between model users in prediction of pesticide leaching using a standard data set. *Pesticide-Science*, 1996, 47: 3, 249-258.

Bunte-D; Pestemer-W; Gottesburen-B; 1992 Application of a simulation model (VARLEACH) for calculation of herbicide distribution in the soil under field conditions. Part 1: influences of spatial variability of soil properties. 16th German conference on weed biology and control, held at Stuttgart-Hohenheim, Germany, on 10-12 March 1992. *Zeitschrift-fuer-Pflanzenkrankheiten-und-Pflanzenschutz-Z.* 1992, 13, 317-325.

Cheah-UanBoh; Kirkwood-RC; Lum-KengYeang; Cheah-UB; Lum-KY; 1997. Adsorption, desorption and mobility of four commonly used pesticides in Malaysian agricultural soils. *Pesticide-Science* 1997, 50: 1, 53-63; based on a communication presented at the COST ACTION 65 6th international workshop 'Pesticides in the soil and the environment', Stratford on Avon, UK, 13-15 May 1996.

Flurry M. 1996. Experimental evidence of transport of pesticides through field soils - A review. *J. Environ. Qual.* 25.

Flurry, M and Fluhler, 1994. *Water Resources Research*, 30, 1945-1954

Groen, KP, (1997). Pesticide Leaching in Polders - Field and model studies on cracked clays and loamy sand. PhD Thesis, Agricultural University of Wageningen, the Netherlands.

Giupponi-C; Bonaiuti-G; Capri-E; Errera-G; Trevisan-M; Re-AAM-del (ed.); Capri-E (ed.); Evans-SP (ed.); Trevisan-M; 1996. Effects of alternative soil tillage systems on the degradation and sorption of herbicides. The environmental fate of xenobiotics. Proceedings of the 10th Symposium Pesticide Chemistry, 30 September-2 October, 1996, Castelnuovo Fogliani, Piacenza, Italy, 1996, 151-161. Facolta grave di Agraria, Universita/grave/ Cattolica del Sacro Cuore, Milan; Italy

Gottesburen-B; Mittelstaedt-W; Fuhr-F; Walker-A (ed.); Allen-R (ed.); Bailey-SW (ed.); Blair-AM (ed.); Brown-CD (ed.); Gunther-P (ed.); Leake-CR (ed.); Nicholls-PH; 1995. Comparison of different models to simulate the leaching behaviour of quinmerac predictively. Pesticide movement to water. Proceedings of a symposium held at the University of Warwick, Coventry, UK, on 3-5 April 1995. 1995, 155-160; BCPC Monograph No. 62; 13 ref. Publisher information: British Crop Protection Council (BCPC); Farnham; UK

Gottesburen-B; Pestemer-W; Bunte-D; Wang-K; Wischniewsky-MB; Zhao-J; 1992 Application of a simulation model (VARLEACH) for calculation of herbicide distribution in the soil under field conditions. Part 2: integration into the expert system HERBASYS. 16th German conference on weed biology and control, held at Stuttgart-Hohenheim, Germany, on 10-12 March 1992. Zeitschrift-fuer-Pflanzenkrankheiten-und-Pflanzenschutz-Z. 1992, Sonderheft 13, 327-336.

Gustafson, DI. (1993). Advantages and potential pitfalls in the use of models for regulatory control of pesticide usage. Brighton Crop Protection Conference

Gustafson, DI. 1999. A tiered approach for ground water study requirements in the United states. First international conference on the behaviour of pesticide in soils, ground water and surface water. June 21 to 22, Darmstadt, Germany.

Jarvis, NJ (1996). Modelling the impact of preferential flow on pesticide fate and mobility. Proceedings of the Cost 66 Workshop, Stratford Upon Avon UK.

Klein, M. (1991). PELMO, Fraunhofer Institute, Scmallenberg, Germany.

Klein M. 1999. Using GIS combined with PELMO to identify areas of different risk categories. Proceedings of the 1st international Conference on the Behaviour of Pesticides in Soils, Grouns and Surface Water June 21 – 22 1999, Darmstadt, Germany.

Letcher R.J., Norstrom R.J. and Bergman A. (1995). Geographical distribution and identification of methyl sulphone PCB and DDE metabolites in pooled polar bear (*Ursus maritimus*) adipose tissue from western hemisphere Arctic and Subarctic regions. Science, Tot. Environ., 160, 409-420.

Mora A, Hermosin, MC and Cornejo, J. Sorption of terbacil by soil as affected by organic/clay mineral ratio. COST 66 workshop on Pesticides in the environment, 13 – 15 may 1996, Stratford-upon-Avon, UK

Nicols, PH, Bromilow, RH, and Addiscott, TM, (1982). Measured behavior of fluometuron, aldicarb and chloride ion in a fallow structured soil. Pesticide Science, 13, 475-483

Partridge, TC and Maud, RR (1987). Geomorphic Evolution of southern Africa since the Mesozoic. S Afr. J. Geol. 90(2), 179-208

Pestemer-W; Gunther-P; Walker-A (ed.); Allen-R (ed.); Bailey-SW (ed.); Blair-AM (ed.); Brown-CD (ed.); Gunther-P (ed.); Leake-CR (ed.); Nicholls-PH; 1995. Use of a pesticide monitoring system (PEMOSYS) for the risk assessment of pesticide leaching potential - concept for post registration activities. Pesticide movement to water. Proceedings of a symposium held at the University of Warwick, Coventry, UK, on 3-5 April 1995. 1995, 351-356; BCPC Monograph No. 62. Publisher information: British Crop Protection Council (BCPC); Farnham; UK

Soil Classification Working Group 1991. Soil Classification, A taxonomic System for South Africa. A Department of Agricultural Development publication.

Trevisan-M; Capri-E; Re-AAM-del; Vischetti-C; Marini-M; Businelli-M; Donnarumma-L; Conte-E; Imbroglini-G; Walker-A (ed.); Allen-R (ed.); Bailey-SW (ed.); Blair-AM (ed.); Brown-CD (ed.); Gunther-P (ed.); Leake-CR (ed.); Nicholls-PH; 1995. Evaluation of pesticide leaching models using three Italian data-sets. Pesticide movement to water. (models VARLEACH, PRZM-2, LEACHP and PESTLA) Proceedings of a symposium held at the University of Warwick, Coventry, UK, on 3-5 April 1995. 1995, 269-274; BCPC Monograph No. 62. Publisher information: British Crop Protection Council (BCPC); Farnham; UK

UNEP. 1998. Proceedings of the subregional awareness raising workshop on persistent organic pollutants (POPs). Inter-Organization Programme for the Sound Management of Chemicals.

Van der Zee, SEATM and Boesten, JJT.(1991). Effect of soil heterogeneity on the leaching of pesticides to groundwater. *Water Resources Research*, Vol 27, no 12 pp 3051-3063.

Vischetti-C. Perucci-P; Scarponi-L; Wagener-A; 1997. Rimsulfuron in soil: effect of persistence on growth and activity of microbial biomass at varying environmental conditions. Special Issue.XII International Symposium on Environmental Biogeochemistry. 1997.

Walker, A and Hollis JM, (1994). Prediction of pesticide mobility in soils and their potential to contaminate surface and groundwater. In *Comparing Glasshouse and field Pesticide Performance II*, ED. HG Hewitt, J Caseley, LG Coppings BT Grayson and D Tyson.

Walker-A; Welch-SJ; Melacini-A; Moon-YH; 1996. Evaluation of three pesticide leaching models with experimental data for alachlor, atrazine and metribuzin. *Weed-Research-Oxford*.1996, 36: 1, 37-47.

Walker A, (1987). Evaluation of a simulation model for prediction of herbicide movement and persistence in soil. *Weed Research*, 27, 143-152.

Weber JB. 1991. Fate and Behaviour of herbicides in soil. *Applied Plant Science*, 5 (1).

Pesticide Drift

Basson, MS, Van Niekerk, PH and Van Rooyen, JA. (1997). Overview of Water Resources Availability and Utilisation in South Africa. Department of Water Affairs and Forestry, Pretoria. 72 pp.

Briggs, P., Raupach, M.R., Cooper, B. and Muschal, M. (1998). Integrative modelling of transport and fate of endosulfan in the riverine environment. Part I: use of observed riverine concentrations to identify contributions of airborne and waterborne transport. CSIRO Land and Water

Cowherd C., Muleski G.E. and Kinsey J.S. (1988). Control of Open Fugitive Dust Sources, EPA-450/3-88-008, US Environmental Protection Agency, Research Triangle Park, North Carolina.

Hofman, V., H. Kucera and M. Berg. (1986). Spray Equipment and Calibration. North Dakota State University Extension Service Circular, 13-AGENG 5-3. North Dakota State Univ., Fargo, ND.

Hwang, S. T., and J. W. Falco (1986). Estimation of Multimedia Exposure Related to Hazardous Waste Facilities. Y. Cohen, Ed. Plenum Publishing

Jury, W.A., W.J. Farmer, and W.F. Spencer. (1984). Behaviour assessment model for trace organics insoil. II. Chemical classification and parameter sensitivity. *J. Environ. Qual.* 13(4) 567-572

Kennedy, I.R., F. Sanchez-Bayo, S.W.L. Kimber, H. Beasley, and N. Ahmad. (1998). Movement and fate of endosulfan on-farm (New South Wales). p. 33837. In *Minimising the impact of pesticides on the riverine environment: Key findings from research with the cotton industry: 1998 Conference*. Canberra. 21B22 July 1998. LWRDRC Occasional Paper 23/98. Land and Water Res. and Development Corp., Canberra, Australia.

Raupach, M.R. and Briggs, P.R. (1998) Integrative modelling of transport and fate of endosulfan in the riverine environment. Part II: Summary Report. CSIRO Land and Water, Consultancy Rep. 50/98.

Raupach, M.R. and Leys, J.F. (1999). The efficacy of vegetation in limiting spray drift and dust movement. CSIRO Land and Water, Tech. Rep. 47/99.

Raupach, M.R., Leys, J.F., Woods, N., Dorr, G. and Cleugh, H.A. (2000). Modelling the effects of riparian vegetation on spray drift and dust: the role of local protection. CSIRO Land and Water, Tech. Rep. 29/00.

Raupach, M.R., Briggs P.R., Ford P.W., Leys J.F., Muschal M., Cooper B., and Edge V.E. (2001a). Endosulfan Transport. I. Integrative Assessment of Airborne and Waterborne Pathways. *J. Environ. Qual.* 30: 714-728.

Raupach, M.R., Briggs P.R., Ahmad N., and Edge V.E. (2001b). Endosulfan Transport: II. Modeling Airborne Dispersal and Deposition by Spray and Vapor, *J. Environ. Qual.* 30: 729-740.

Ross M.A. and Lembi C.A. (1985). *Applied Weed Science*. Burgess Publishing Company, Minneapolis, MN.

Raupach, M.R., Woods, N., Dorr, G., Leys, J.F. and Cleugh, H.A. (2001c). The entrainment of particles by windbreaks. *Atmos. Environ.* 35, 3373-3383.

Teske, M.E., Bowers, J.F., Rafferty, J.E., and Barry, J.W., (1993). FSCBG: An Aerial Spray Dispersion Model for Predicting the Fate of Released Material Behind Aircraft, in *Environmental Toxicology and Chemistry*, Vol. 12, pp. 453-464.

Teske M.E., J.W. Barry and H.W. Thistle, (1994). *Aerial Spray Drift Modeling*. @ Environmental Modeling Vol. II. P. Zanetti (ed.), Computational Mechanics Publications, Boston.

Teske, M. E., S. L. Bird, D. M. Esterly, S. L. Ray, and S. G. Perry. (1997). A User's Guide for AgDRIFT 1.0: A Tiered Approach for the Assessment of Spray Drift of Pesticides, Technical Note Number 95-10. Continuum Dynamics, Inc., Princeton, NJ.

Teske, M.E., Curbishley, T.B., (1990). Forest Service Aerial Spray Computer Model FSCBG 4.0 User Manual, Technical Report Number 90-06, Continuum Dynamics, Inc., Princeton, NJ.

Teske, M. E., H. W. Thistle and B. Eav, (1998). New Ways to Predict Aerial Spray Deposition and Drift, in *Journal of Forestry* 96(6):25

11. ACKNOWLEDGEMENTS

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DISCUSSION PAPER 3:

SEDIMENTS AND METAL POLLUTION AND RELATED ACTIVITIES

by

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SEDIMENTS AND METALS

1. INTRODUCTION

1.1 SEDIMENT AS A NON-POINT SOURCE POLLUTANT FROM AGRICULTURE

Sediment is the most widespread pollutant of surface water and has negative off-site effects on most water users. Sedimentation of streams and impoundments reduces the storage capacity of rivers, enhances the risk of flooding, blocks irrigation canals, shortens the design life of reservoirs and reduces recreational opportunities. Increased turbidity and sedimentation affects the functioning and productivity of the aquatic environment, by decreasing light penetration for aquatic plants, smothering aquatic habitat, stressing filter-feeders and interfering with predator's feeding. Sediment is not only a widespread pollutant of surface water, but is also the most important medium for transport of phosphorus, heavy metals, toxic substances and pathogens from agricultural land to surface waters. Many pathogens, pesticides and their metabolites, heavy metals and major plant nutrients such as phosphorus are readily adsorbed on the soil particles and transported as sediment-bound pollutants.

Land disturbances associated with agriculture are one of the major sources of sediment. Most of the land area in South Africa is utilised for agricultural activities which have been implicated as a major source of sediments. It is therefore necessary to assess the contribution that the different agricultural activities make towards the delivery of sediments to surface waters, to devise the approaches through which these can be controlled and to determine and predict the effect that control measures will have on reducing the production and delivery of sediment to watercourses. An understanding of soil erosion and sediment movement is, therefore, fundamental to an appreciation of catchment water quality associated with sediments and chemicals that most likely are adsorbed to sediments. After hydrology, understanding sediment production and delivery from agricultural land to the watercourses is essential for non-point source assessment (Baldwin, Mitchell and Olley, 2002; Pegram and Gorgens, 2001; Wasson, 1998).

This position paper was prepared to present a reconnaissance review of the knowledge base regarding the production and delivery of sediments from agricultural land to surface waters. The transport processes of sediment-bound non-point source pollutants were also discussed where appropriate.

1.2 METALS AS NON-POINT SOURCE POLLUTANTS FROM AGRICULTURE

Heavy metals are elements that have a high density and belong largely to the transition group of the periodic table. The elements that are generally considered include arsenic, cadmium, cobalt, chromium, copper, mercury, molybdenum, nickel, lead, selenium, vanadium and zinc (De Vries, Römken, van Leeuwen and Bronswijk, 2002).

Heavy metals occur naturally in small quantities in surface water resources as a result of geological weathering. High concentrations of heavy metals are found in waterbodies as a result of human activities. In the majority of cases in South Africa, heavy metals reach rivers, wetlands and estuaries from point sources. Mining activities are the main heavy metal polluters, whilst other polluters include industries that use or manufacture metals. Urban runoff can also contain large quantities of heavy

metals (Snaddon, 1999; Dallas and Day, 1993). Heavy metal pollution of water resources from agricultural land is restricted to point sources where land is utilised for confined animal feeding operations, and for the disposal of industrial effluents and solid wastes and of sewage sludges (De Vries *et al.*, 2002; Chang and Page, 2000; Chartres and Webb, 1998). Discussion of these point sources is beyond the scope of this study.

The non-point sources of heavy metals in agriculture are from irrigation due to trace amounts in irrigation water; application of fertilizers and soil ameliorants which contain heavy metals as impurities or supplements; and past uses of heavy metal-based insecticides and herbicides. Despite the inputs from various sources, there is in recent times a depletion of several metals essential for plant growth from dryland soils due to the application of high grade fertilizers and organochlorine and organophosphate pesticides (De Vries *et al.*, 2002; Chang and Page, 2000). Herselman and Steyn (2001) have found a depletion of several metals essential for plant and animal growth in dryland soils of the Mpumalanga Province, especially for the resource-poor farm lands. They also found that the heavy metal contents are much higher in the irrigated land than in the dryland cropland, rangeland and resource-poor farm lands.

The non-point source contribution of heavy metals from agriculture to surface waters is low since the inputs of heavy metals to dryland agriculture are minor and most of the applied heavy metals to agricultural land are strongly bound to soil particles (Chadwick and Chen, 2002). Heavy metals as an agricultural non-point source pollutant are not further discussed since they are not a major threat to water quality such as salinisation, eutrophication and sediments (DWAF, 1986), and various reactions and mechanisms were applicable for a range of elements.

2. BRIEF OVERVIEW OF EXTENT OF SEDIMENTS

Many South African rivers carry a naturally high suspended-solids load and permanently turbid water is noticeable in rivers in the former Transkei area, KwaZulu-Natal and in many reservoirs around the country (Snaddon, 1999). Sediment load in southern African rivers is highly variable, not only the daily loads of rivers, but even on an annual basis (Rooseboom, 1992).

The most important source of data on sediment yield consists of the reservoir re-surveys which are performed on a regular basis by the Department of Water Affairs. Recorded suspended sediment load records are available for a number of South African river gauging stations which were operational for limited periods of time after 1928 (Rooseboom, 1992). Rooseboom, Verster, Zietsman and Lotriet (1992), produced a revised sediment yield map for South Africa, which was based on records of sediment loads of reservoirs and rivers around the country. The map indicates that much of the interior and east coast falls within the moderately high to high sediment yield classes.

On a global basis, documented suspended sediment yields include minimum values well below $2.0 \text{ t/km}^2/\text{yr}$, while values in excess of $10,000 \text{ t/km}^2/\text{yr}$ have been reported for several rivers (Walling, 1994). The most quoted South African sediment yield estimates are Midgley's (1952) figure of 363 million tonnes ($300 \text{ t/km}^2/\text{yr}$), Schwartz and Pullen's (1966) value of 233 million tonnes ($190 \text{ t/km}^2/\text{yr}$) and Rooseboom's (1978) estimate of 100-150 million tonnes ($82\text{-}122 \text{ t/km}^2/\text{yr}$), based on the sediment yield of main rivers. For comparison the estimated average for Australia is $270 \text{ t/km}^2/\text{yr}$, and the mean for Africa is $750 \text{ t/km}^2/\text{yr}$ (Garland, Hoffman and Todd, 1999).

The World Map on Status of Human-Induced Soil Degradation (UNEP/ISRIC, 1990) shows that water erosion, largely induced by agricultural practices, affects more than 70% of South Africa's surface area. Most research data on soil erosion has been obtained from runoff plots, and it is important to realise that extrapolation from runoff plots to catchments is problematic, and direct proportional conversions will undoubtedly result in over-estimations. Only a fraction, and often a rather small fraction (*0.05 is frequently cited*), of the sediment eroded (soil loss) within a catchment reaches the catchment outlet and is presented in sediment yield. Deposition and temporary or permanent storage may occur on the slopes, particularly where gradients decline downslope, at the base of a slope, in swales, on floodplains and in the channel itself (Walling, 1994). However, soil loss data (gross erosion) on various agricultural land uses can give valuable insights on the non-point source impact of erosion. Measured values of soil losses from undisturbed veld range from 0.02 t/ha/yr to 0.75 t/ha/yr ($2\text{-}75 \text{ t/km}^2/\text{yr}$). With few exceptions, soil losses from agricultural land uses fall between the extremes of 3 and 72 t/ha/yr ($300\text{-}7200 \text{ t/km}^2/\text{yr}$). Of crops offering poor protection, maize, with measured values commonly around 25 t/ha/yr ($2500 \text{ t/km}^2/\text{yr}$), but on occasion attaining more than 60 t/ha/yr ($6000 \text{ t/km}^2/\text{yr}$) is probably the worst (Garland, Hoffman and Todd, 1999). Based on hydrological data collected from sugarcane plots, Schmidt, Smithers, Schulze and Mathews (1998) reported that the average soil loss for sugarcane ranged from 21 t/ha/yr ($2100 \text{ t/km}^2/\text{yr}$) to less than 2 t/ha/yr ($200 \text{ t/km}^2/\text{yr}$).

The availability of information on the extent of surface water degradation in South Africa, such as on sediment, is extremely variable. Information for the former homelands and for communal areas is scarce (Snaddon, 1999). Although there are a number of perceptions and ideas relating to the non-point source impact of erosion, they are mostly qualitative and geographically limited. The monitoring of a few benchmark sites representing a nested system of runoff plot-hillslope-small catchment-larger catchment on a regular basis, with reliable data acquisition and storage, would be valuable to quantify sediment production, sediment delivery to watercourses and sediment transport from smaller catchments to large catchments, and also the trends and changes taking place.

3. DESCRIPTION OF ORIGINS AND SOURCES

The major sources of sediment are natural erosion and land disturbances associated with urbanisation, mining, silviculture and agriculture (Table 3.1). Although erosion rates associated with surface mining and urbanisation are much higher than from agricultural sources, these activities only occupy a small percentage of the land attributed to agricultural activities (Elliot, Ward, Lane and Laffin, 1989).

Table 3.1 Major sources of sediment

NATURAL	Erosion of landscapes Erosion of stream courses
URBANISATION	Construction of buildings Construction of infrastructure Storm water Land clearing
MINING	Quarrying of stone and gravel for roads, buildings, etc. Extraction of gravel from stream beds Open-cast mining Mine dumps
MILITARY	Military manoeuvres Vehicle testing Off-road driving courses
SILVICULTURE	Cultivation or draining prior to planting Construction and use of roadways Harvesting
RECREATION	Water-based activities Hiking trails Camping sites Off-road driving
AGRICULTURE	Cultivation Grazing Irrigation Watering points

3.1 NATURAL EROSION

Although all land is subject to geomorphological processes, notably weathering and erosion, the interest here is on the acceleration of sediment generation and delivery above natural background rates, especially the suspended wash load of finer particles and aggregates. In order to study the impact of agriculture on sediments, it is important to separate natural background rates from the accelerated rates due to agriculture. For instance, bank erosion by fluvial processes, such as meander migration, is clearly different from damage to stream banks by animals. When sediment yield is determined/predicted, one should bear in mind that the natural background rates must be subtracted in order to study the impact of agriculture.

3.2 AGRICULTURE

Generation of sediment by agriculture originates from damage to the soil by ill-timed or ill-suited use of the land.

3.2.1 Cropland

Accelerated erosion occurs by attempting to farm slopes that are too steep, cultivating up-and-down hill, continuous use of the land for the same crop without fallow or rotation, inadequate use of fertilizers, compaction of the soil through the use of heavy machinery and pulverization of the soil when trying to create a seedbed. The soil is most vulnerable to accelerated erosion during preparation for sowing and planting, before the establishment of the crop's canopy and after harvest of some crops. Many farmers prepare land by tilling or ploughing their fields to produce a smooth planting surface. This process, however, pulverizes the soil near the surface and creates a soil surface that is very vulnerable to erosion, as well as a compacted layer at plough depth which reduces infiltration and results in increased runoff.

3.2.2 Rangeland

Since grass provides a dense cover close to the soil surface, it is a good protector of the land against erosion. Erosion problems arise only when this cover is removed through over-grazing, although they can be exacerbated by drought and excessive burning. Erosion also results from traffic by animals, especially under high stocking rates, which causes much soil disturbance at gateways, tracks, verges, hedges and banks of watercourses.

3.2.3 Forestation

The most important erosion problems are associated with cropping of the trees for firewood, destruction of the trees and surrounding shrub and ground cover by grazing, and logging operations. The main erosion problems in logged areas are associated with skid trails and roads which are frequently areas of bare compacted soil (Morgan, 1995).

4. DESCRIPTION OF PROCESSES

4.1 SCALE AND EROSION PROCESSES

4.1.1 Process Scale

The process scale is the scale that natural phenomena exhibit. These scales are not fixed, and may in fact vary according to the process involved. Many processes in natural systems exhibit one or more preferred scales. There is an intrinsic relationship between space and time scales. Processes operating at detail spatial scales operate at higher frequencies, while processes occurring over broader spatial scales operate at lower frequencies (Jewitt and Gorgens, 2000).

Soil erosion phenomena occur at a range of space and time scales. The magnitude and processes of erosion are different at the various scales. Some of these relate to the dominant dynamic interactions which create patterns at different spatial scales, and the ways in which these different scales interact (Kirkby, Imeson, Bergkamp and Cammeraat, 1996; Poesen, Boardman, Wilcox and Valentin, 1996).

4.1.2 Scale and Hierarchy

A hierarchy is a formal organisation of various spatial or temporal sizes or levels graded from detail to general. According to Hierarchy Theory, nature can be partitioned into "naturally occurring" levels that share similar time and space scales, and that interact with higher and lower levels in systematic ways. One end of the spectrum is characterised by high perspective and low detail, and the other end, by high detail and low perspective. Movement towards a lower level results in details becoming clearer, while assemblages, patterns, and relationships formed by these become less discernible. Movement in the opposite direction, towards a higher perspective region, results in the opposite effect: assemblages, patterns and relationships emerge while the detailed processes that form them become more obscure. In other words, look one level down a hierarchy for more detail and one level up for more context. Many hierarchies developed for natural systems are spatially nested, i.e. a system at one level forms the environment or context for systems making up the lower levels of the hierarchy (Jewitt and Gorgens, 2000).

A spatially nested hierarchy suggested for erosion studies is the global, national, regional, catchment, hillslope, land facet and microscopic scales. The global, national and regional scales are not discussed since these scales are not relevant to the scope of this study.

4.2 EROSION PROCESSES

Soil erosion by water is a three-phase process, involving detachment of individual particles from the soil mass, their transport by impacting raindrops and running water, and deposition when sufficient energy is no longer available to transport the particles.

The severity of erosion depends upon the quantity of material supplied by detachment and the capacity of the eroding agents to transport it. Where the agents have the capacity to transport more material than is supplied by detachment, the erosion is described as detachment-limited. Where more material is supplied than can be transported, the erosion is transport-limited (Morgan, 1995). According to Rooseboom (1992), *sediment availability rather than transporting capacity proves to be the limiting factor in determining sediment yields in most southern Africa rivers*. This is attributed to the fact that the hydraulic transporting capacity in southern African rivers is generally sufficient to carry the predominantly fine sediments in the rivers.

4.2.1 Catchment Scale

The catchment is the land surface which contributes water and sediment to any given stream network (Rowntree and Wadeson, 1999). Apart from the smallest unbranched headwater catchments, all catchments are made up of a hierarchy of smaller catchments. The spatial extent of catchments varies from areas of much less than 1 km² up to over 10⁵ km²; consequently, the different processes that may contribute to sediment yield, and rates of change of a catchment are not consistent across different catchment sizes.

The relevant erosion processes that occur at the catchment scale are interrill and rill erosion, gully erosion, stream channel erosion and flood plain scour. The routing of sediment through the catchment is an important process at the catchment scale. Gully erosion, and interrill and rill erosion were discussed at the hillslope and land facet scales respectively (Wasson, 1998).

Stream channel erosion and flood plain scour

Stream channel erosion refers to erosion from the banks and bed of the stream channel due to scour induced by stream flow. Stream bank scour most often closely resembles rill erosion, which is often followed by bank collapse due to side slope instability. Channel erosion interacts strongly with the incoming sediment load. When inflow sediment is reduced from hillslopes, the stream channel and previously stored sediment in the channel may erode with large sediment yield rates continuing for several years (Foster and Meyer, 1977).

The sediment transport in stream channels may occur according to different transport modes. They are mass transport, suspended load and bedload. The transport of particles as bedload is mainly by tractive and gravitational forces. The re-entrainment of eroded materials from the bed and banks of a channel is caused by the tractive force of streamflow in the channel. Saltation (bouncing movement of small grains of particles in water with rebound along the stream bed) plays an important role in re-entrainment (Lenzi, Agostino and Billi, 1999).

Under conditions of excessive erosion and landscape instability, stream channels that have a relatively smooth, concave-upwards long profile can develop into valley floor gullies that are characterized by steep banks and large sediment loads that display very erratic behaviour so that relationships between sediment discharge and runoff are frequently poor. The processes operating in valley floor gully development are the same as for valley side gullies, which is discussed at the hillslope scale (Morgan, 1995).

Flood plain scour may also occur in large catchments where alluvial deposits become larger with distance downstream. Flood plain scour is the detachment and transport of soils on flood plains during flooding and the processes are much the same as for stream channel erosion (McLaughlin, Kookana, Donnelly and Wasson, 1998).

Spatial distribution of erosion

At the small catchment scale (< 10 km²), the several nested levels of mosaic patterns of hydrological response units play a dominant role in the initiation of runoff, and sediment production and delivery in a catchment. For a storm to initiate catchment-scale runoff and erosion it must overcome the spatial arrangement and threshold values of all the nested hydrological response units at the hillslope scales. Widespread runoff and erosion at the catchment scale therefore requires prolonged or larger magnitude storms. In catchments where variable source area hydrology is a dominant process, hydrologically active areas (HAA) are more susceptible to producing sediments. Runoff from most of the HAA quickly moves downhill to stream channels, thus providing a potential direct hydrological and erosion link between the hillslopes and stream channels. Recognizing the existence of HAAs limits

the scope of catchment-scale water quality problems to only those areas where HAAs coincide with land uses that potentially contribute sediments (Walter, Walter, Brooks, Steenhuis, Boll and Weiler, 2000; Endreny and Wood, 1999).

Sediment delivery

In small catchments the source of sediment is usually obvious, but in a large catchment the source of downstream sediment is not at all obvious. Not only is a large catchment more complex than a small one, there are several major problems related to sediment delivery in linking on-farm erosion with downstream sedimentation.

The first stems from the fact that only a fraction of the sediment eroded within a catchment reaches the catchment outlet and is represented in the sediment yield. Deposition and temporary or permanent storage may occur on the hillslopes; at the base of a hillslope; in swales; on the flood plain; and in the channel itself. The relative magnitude of this loss tends to increase with increasing catchment size (Walling, 1994). In the headwater catchments the hillslopes and stream channels are usually directly connected, and the channels are steep which can remove most sediment delivered to them from adjacent hillslopes. Therefore, sediments eroded from the hillslopes of headwater catchments are likely to affect water quality. Further downstream, hillslopes are separated from channels by flood plains, terraces, ditches and other depressional areas that intercept sediment from interrill and rill eroded areas and discontinuous gullies leaving the hillslopes. Consequently, sediment transfer through a catchment is unlikely to be continuous. In these catchments, continuous gullies, stream channel erosion and flood plain scour probably contribute greater amounts of sediment to rivers than interrill and rill erosion and discontinuous gullies, due to eroded sediments directly entering the waterway. In many catchments typical of dryland farming regions, a pulse of sediment derived from agricultural land in catchment headwaters is attenuated during its passage through the catchment. The magnitude of yield increase is therefore greatest in the cultivated or grazed headwaters and least in downstream river sections.

In some landscapes, active gullies are important sources of sediments and are likely to be the major sources of sediment in upland catchments until they have equilibrated. Not only are gullies important sources of sediment, but, because the sources and mode of transport are adjacent to the watercourses, the sediment delivery is higher than for an ungullied or lightly gullied catchment (Wasson, 1998).

A second problem is the temporal discontinuity that may be involved in sediment delivery. Sediment eroded at one location may be stored temporarily and subsequently remobilised several times before reaching the catchment outlet. Sediment transported out of a catchment may therefore reflect the recent history of erosion and sediment delivery rather than contemporary erosion in the headwater areas. When soil conservation measures are implemented to reduce upland erosion, a significant reduction in sediment yield may take decades, because sediment stored in the valley deposits is then remobilised (Walling, 1994).

Rooseboom (1992) has derived equations which describe the sediment transport-carrying capacity of runoff in catchments for southern African conditions which do take account of sediment availability in addition to other factors frequently used in transport models such as catchment slope, rainfall intensity and runoff. He concluded that by the time the transported sediments reach main river courses, the carrying capacity tends to be far greater than that which is required to transport the sediment load. *It proved that the availability of sediment for transport plays the dominant role in determining sediment loads in southern African rivers.*

Transport of sediment-bound pollutants

Sediment-bound pollutants present within the aquatic system can be delivered from hillslopes and stream banks and beds. The delivery of sediment-bound pollutants from hillslopes is discussed in Section 4.2.2.3. The sources of sediment-bound pollutants in large catchments are not always known due to the problems related to sediment delivery in linking on-farm erosion with downstream sedimentation (Section 4.2.1.3) and the complex interplay between the sediment and pollutants when transported through an aquatic system (Herpe, Troch, Quinn and Anthony, 2002).

Sediment deposition may be an important pathway for the removal of a pollutant from the water column. Conversely, sediment re-suspension and transport can be an important mechanism for the movement of the pollutant through an aquatic system. Suspended and bedload sediments are important sinks for the removal of dissolved pollutants from aquatic systems. Abiotic pathways for the incorporation of dissolved pollutants into sediments include adsorption and precipitation/co-precipitation reactions. However, sediment-pollutant interactions in the aquatic system often go beyond simple sorption and precipitation processes since biotic and abiotic processes associated with sediments may transform the pollutant.

Sorption of pollutants on to sediments followed by sediment deposition and then slow re-release of the pollutant back into the water column can be seen as buffering the potential impact of episodic pollution events. However, this buffering also serves to prolong the impact of a given pollutant when the removal of pollutant inputs into streams often does not result in a noticeable decrease in pollutant concentrations, because of re-release of the pollutant from the sediments. Therefore, sediments can be seen as temporally shifting pollutant impacts (Baldwin *et al.*, 2002).

Factors

Hydrology

The erosivity of floodwaters in stream channels depends on their volume and velocity which are in turn influenced by rainfall intensity and duration in the catchment; catchment size, topography and vegetation; infiltration and water storage capacities of soils in the catchment; stream channel size; and gradient and surface topography of the flood plain (McLaughlin *et al.*, 1998). The effect of hydrology on the spatial distribution of erosion was discussed at the hillslope scale.

Geology

The geology of a catchment exerts a fundamental influence on both runoff and erosion. Rock type and structure both have a direct influence on the potential for groundwater storage in a catchment and therefore on the partition of runoff between storm flows and base flows. This may also be reflected in the drainage density. A permeable rock with high groundwater storage potential has a few large streams with wide interfluvies, in contrast to less permeable rocks which will tend to have a high drainage density. Depending on climatic influences, a large proportion of the drainage network of less permeable rocks will serve only to carry storm runoff, so that many channels will be ephemeral.

Sediment yield is also related to geology through such factors as the weathering rate and the size distribution of the weathered products. For example, in the Eastern Cape a distinction can be made between erodible soils developed on the silts and mudstones of the Beaufort Series which give rise to fine textured dispersive soils and the dolerites which give rise to well structured clay soils with a lower erodibility. The quartzitic Table Mountain sandstones weather more slowly, producing coarse textured soils with very little silt and clay, thus accounting for the clear waters and widespread distribution of sand bed rivers in the lowlands of the Western Cape (Rowntree and Wadeson, 1999).

Topography

Topography affects the continuity between upland erosion and downstream sedimentation as it determines the deposition and temporary or permanent storage of colluvium on hillslopes or as alluvium in swales and stream channels and on the flood plain (Wasson, 1998).

Agricultural practices

The effect of agricultural activities on water quality associated with sediments will vary with position of the farm in a catchment because of the inherent variation both in landform configuration, in the erosion rates between landforms and sediment routing from a hillslope to a specific point in a drainage system. Most of the variance in sediment yields from small upland catchments is the result of variation in agricultural land use. However, once gullies have developed, it can mask the effect of variation in agricultural land use on sediment yields. For catchments larger than 1000 km², it is unlikely that agricultural enterprise is a useful explanatory variable, given the powerful attenuation of sediment transport in large catchments by storage (Wasson, 1998).

4.2.2 Hillslope Scale

A small catchment consists of a stream channel which is linked with hillslopes. A hillslope is an area extending from the watershed of a catchment down to the stream channel and represents the spatial variation of topography, soil and land management patterns along the hillslope.

The erosion processes that occur at the hillslope scale are, interrill and rill erosion, and gully erosion. Interrill and rill erosion were discussed at the land facet scale. The spatial distribution of hydrological response units and hence the spatial connectivity of runoff-producing areas is an important process determining the spatial extent of the erosion processes along the hillslope and the major sediment delivery processes to the stream channel.

Gully Erosion

Gullies typically form in hillslope hollows and on lower slopes where the topography tends to concentrate overland flow into preferred pathways of flow. Gullies may be continuous, that is they discharge into the stream at the bottom of the slope, or discontinuous, fading out into a depositional zone and not reaching the valley floor.

Gully formation can result from incision by overland flow, from subsurface flow and by surface collapse from piping. Gully erosion proceeds by a complex of processes which may include: scour by concentrated runoff, headward erosion of gullies, gully widening, and dispersion in subsoils around seepage channels, followed by tunnel formation, collapse of the overlying soil into the tunnel, and surface gully flow (Morgan, 1995; Bocco, 1991).

A gully that forms by enlargement of a rill may have no headscarp because it forms in non-cohesive materials without a resistant capping. This type of gully, called ephemeral gullies, was a master rill which has so deepened (> 30 cm) and widened its channel that it is classed as a gully. Ephemeral gullies usually increase in width and depth downslope as the runoff increases since processes for ephemeral gully development are the same as for rills discussed in Section 4.2.3.2 (Morgan, 1995; Selby, 1993).

Gullies with headscarps develop where non-cohesive materials are capped by resistant soil and the eroded material is transported away from the base of the headscarp. The development of gullies with headscarps proceeds by a complex of processes. In the first stage small depressions or knicks form on a hillslope in much the same way as an incipient rill (Section 4.2.3.2) is formed. Erosion is concentrated at the heads of the depressions where near-vertical scarps develop over which supercritical flow occurs. The supercritical flows are causing the formation of a waterfall and plunge

pool leading to undercutting and slumping of the gully head. Tension cracks in the gully head can also develop due to gravitational forces and can lead to gully head failure. Some soil particles are detached from the scarp itself but most erosion is associated with scouring at the base of the scarp which results in deepening of the channel and the undermining of the headwall, leading to collapse and retreat of the scarp upslope. This process is accentuated by subsurface seepage. The rate of downcutting, particularly in the head area, is very rapid until the channel reaches the groundwater zone or bedrock. Thereupon, downcutting ceases but the gully continues to expand by sapping, undermining and collapse of the headwall and side walls. In this stage, upslope runoff contributes little, if anything, to the gully growth whereas localized concentrations of subsurface seepage and/or groundwater enhance headwater bifurcation. Sediment is also produced further down the gully by bank erosion. This occurs partly by the scouring action of running water and the sediment it contains and partly by slumping of the banks following saturation during flow. Between flows, sediment is made available for erosion by weathering and bank collapse (Wijdenes, Poesen, Vandekerckhove and Ghesquiere, 2000; Morgan, 1995; Selby, 1993; Bocco, 1991).

A third class of gullies is those which are initiated by subsurface flows in natural pipes, with subsequent collapse of the roof of the pipe. In some hillslopes, rainstorm water infiltrates rapidly down cracks and super-saturates a relatively permeable horizon in the subsoil. Lateral seepage may be fast enough to move soil particles and develop a channel, or, if the soil has dispersible clays, these may lose aggregation. Movement of water through subsurface cracks and voids is slow until water breaks through the soil surface further down the slope, and rapid flow can then work headwards within the soil and form a gully or enlarge a pipe. When heavy rain provided sufficient flow to flush out the soil in these, the ground surface subsided, exposing the pipe network as gullies (Beckedahl, 1998; Selby, 1993).

Over time, the active growth of gullies causes the channel slope to be reduced, promoting a consolidation phase as the gully stabilizes, the channel fills in, the sides and head wall become flatter and vegetation regrows. Deposition steepens the slope again and triggers a new phase of gullying. Thus gullies pass through successive cycles of erosion and deposition. It is not uncommon for the head of a gully to be extremely active whilst the lower section of the gully is stabilizing or for gullies to contain a sequence of alternating stable and unstable sections (Morgan, 1995).

Spatial distribution of erosion

The combination of a non-uniform spatially distributed vegetation, irregular terrain and complex pedological patterns along a hillslope results in a situation whereby the hydrological response of hillslopes to rainfall events may be spatially non-uniform with overland flow occurring in localised areas. The development of continuous hydrological pathways increases the vulnerability to erosion and is critical in determining sediment delivery from hillslope erosion to the channel system.

Surface runoff from source areas which are spatially isolated and upslope of the channel will be re-absorbed by the surrounding drier areas which act as sinks for overland flow and transported sediment. During dry weather conditions a mosaic of areas with contrasting hydrological response exists and erosion areas are spatially isolated and unconnected. During wet weather conditions, the increased spatial continuity in hydrological pathways results in widespread runoff and erosion. Storm-runoff contributing areas commonly develop first alongside stream channels and in concavities on the hillslope (Walter *et al.*, 2000; Endreny and Wood, 1999; Selby, 1993).

Delivery of sediment-bound pollutants

The hillslope scale encompasses the major pathways and transport processes on the delivery of sediment-bound pollutants to watercourses. The pathways include both surface movement of pollutants associated with colloidal and particulate transport, and subsurface movement associated

with solid phases. Surface transport of pollutants is associated with sediment transport from interrill areas to rills by broad shallow overland flow, and sediment transport in rills and ephemeral gullies by concentrated flow (McLaughlin *et al.*, 1998). Fine colloidal-sized particles may be particularly efficient at transporting pollutants through subsurface pathways, because they have high capacities to adsorb pollutants but are not easily stabilized in the soil profile (Leinweber, Turner and Meissner, 2002). In soils with macropores and pipe gullies, sediment-bound pollutants are rapidly transported through these preferential flow pathways (Chadwick and Chen, 2002).

The runoff generation mechanisms, spatial distribution of hydrological response units and the development of continuous hydrological pathways (Section 4.2.2.2) are critical in determining the delivery of sediment-bound pollutants from agricultural land to the watercourses, since they determine the dominant transport mechanisms and pathways of pollutants through the landscape (Nash, Halliwell and Cox, 2002). Gully erosion can also contribute significant amounts of sediment-bound pollutants to the watercourses which can dominate river fluxes of pollutants delivered by sediments (McLaughlin *et al.*, 1998). Chemical reactions may occur between the pollutants in soluble phase and on the sediment at any stage during transport from the soil to the watercourses (Haygarth and Jarvis, 1999).

Factors

Climate

Gully growth takes place episodically most often during infrequent large storms (Bocco and Garcia-Oliva, 1992).

Soil

At low moisture contents the soil at gullies behaves as a solid and fractures under stress but with increasing moisture content it becomes plastic and yields by flow without fracture. With further wetting, the soil will reach its liquid limit and start to flow under its own weight.

Gully development is favoured by a soil with a zone of soil moisture concentration within the soil profile, a zone of preferential subsurface water movement in response to a hydraulic head, a high percentage of swelling clays and associated cracking of the soil profile in dry periods; an erodible layer above a relatively impermeable layer in the soil profile; and a dispersible soil layer (Beckedahl, 1998; Morgan, 1995; Selby, 1993).

Topography

Slope length and form will determine the severity of erosion through the type of erosion that will dominate, and the probability for deposition. On concave slopes most erosion takes place on the upper third, while on convex and straight slopes most erosion takes place about three quarters of the way down the slope. However, not all concave slopes are depositional. The accumulation of overland flow and soil moisture in concave downslope positions may lead to severe rilling and gully. Convex slope topography produces relatively low sediment yields, because of divergent flow lines. Concave slopes produce low sediment yields, but concave slopes with active gullies produce the greatest quantities of sediment (Morgan, 1995).

In contradiction with interrill and rill erosion, gully erosion cannot be correlated positively with slope steepness. The prevalence of gentle slopes in concave downslope positions or at terraces along the hillslope allows both the deposition of material susceptible to erosion, and for higher initial soil moisture content due to subsurface flows (Bocco and Garcia-Oliva, 1992).

Vegetation

Vegetation affects both gully initiation and gully growth. Severe degradation of vegetation results in the critical value for shear stress of water flows being lowered to such an extent that storms with a greater frequency of occurrence can initiate gully erosion. Furthermore, any local disturbances such as cattle tracks can initiate gully formation (Selby, 1993).

A vegetation cover should contribute to slope stability of gullies as a result of evapotranspiration producing a drier soil environment so that a higher intensity and longer duration rainfall is required to induce slope failure compared with an unvegetated slope. However, if subsurface flows are very active, as occurs in gently sloping areas, promoting infiltration may lead to increase gully erosion (Morgan, 1995; Bocco and Garcia-Oliva, 1992).

Hydrology

Initial soil moisture, connectivity between runoff source areas and the development of continuous hydrological pathways depends on the interaction between agricultural management practices and landscape characteristics. High rainfall intensity, bare soils and surface crusts will tend to promote widespread runoff and erosion. Partial area flow conditions will be encouraged on steeper sloping lands and where soils are strongly duplex, or have shallow sandy strata overlying impermeable horizons, as these characteristics will promote subsurface flow (Hook, 1998; McLaughlin *et al.*, 1998).

Soil saturation plays an important role in the occurrence of ephemeral gullies, which were found exclusively in areas of soil saturation. Soil saturation also plays an important role in the occurrence and extension of classical gully erosion. When saturation occurs, the relatively impermeable B horizon supports lateral flow in the A2 horizon. Where a gully intersects this flowpath there is seepage from the A2 horizon on the face of the gully wall which may result in sapping (outward flow of the fluidised A2 horizon material), or erosion of the highly dispersible B horizon (Armstrong, Mackenzie and Edwards, 1995).

4.2.3 Land facet scale

A hillslope consists of land facets along the hillslope. A land facet is an area of homogeneous topography, soil and land management.

The land facet scale represents the combined processes of rill and interrill erosion. On the upper parts of hillslopes, particularly on those of convex form, interrill erosion is the dominant erosion process. Rills are initiated at a critical distance downslope where overland flow becomes channelled and experience has shown that 5 m is about the minimum slope length that will adequately represent a rill system (Morgan, 1995).

Interrill erosion

Interrill erosion is the detachment and transport of soil by raindrop impact (rainsplash erosion) and shallow overland flow (Lal and Elliot, 1994). Detachment on interrill areas is primarily induced by rainsplash erosion. Transport on interrill areas is primarily by broad shallow overland flow (Nearing, Lane and Lopes, 1994).

Rainsplash erosion encompasses the processes of surface sealing, raindrop detachment and splash transport (Mutchler, Murphree and McGregor, 1994). The detachment of soil particles represents a failure of the soil by the combined mechanism of compression and shear under raindrop impact. Once a critical depth has been reached (about three times the diameter of a raindrop), detachment decreases exponentially with increasing water depth.

Overland flow is generated when surface depression storage and either soil moisture storage or the infiltration capacity of the soil are exceeded (Morgan, 1995). A soil particle entering overland flow due to rainsplash erosion may be transported downstream in one of three modes. If it is small or of low specific gravity, it may move as suspended load. If larger or of higher specific gravity, it may fall to the underlying surface (bed) and, if the flow has sufficient velocity, move as bed load. However, those particles not entrained remain on the bed until lifted back into the flow by impacting raindrops (Morgan, 1995).

Rill erosion

Rills are initiated at a critical distance downslope where unconcentrated overland flow becomes concentrated flow paths which can lead to micro-channels when the critical shear velocity of runoff exceeds the shear strength of the soil. Once rills have been formed, their migration upslope occurs by the retreat of the headcuts on the steep banks at the top of the channel. Downslope extension of the rill is controlled by the shear stress exerted by the flow and the strength of the soil.

When soil particles have been detached and entrained within the flow, they will be transported until such time as deposition occurs. Sediment transport depends on flow volume and sediment concentration. The actual sediment concentration in the rill and, therefore, the erosion may vary considerably. This is because the rill has to adjust continually for pulse influxes of sediment due to wash-in from interrill flow, erosion and collapse of the head wall and collapse of the sidewalls (Morgan, 1995).

Enrichment of sediment-bound pollutants

During interrill erosion the finer-sized soil particles of the soil mass are selectively eroded and then preferentially transported from the interrill areas to rills. This is often associated with the preferential transport of finer-sized soil particles in rills. The effect of preferential detachment and transport of the finer-sized soil particles leads to pollutant enrichment. The adsorption of the pollutant during the transport process further increases the pollutant enrichment. This has led to the determination of enrichment ratios, calculated as the ratio of the concentration of pollutant on the sediment (eroded soil) to that in the soil mass (McDowell, Sharpley, Kleinman and Gburek, 2002; Haygarth and Jarvis, 1999).

Factors

Climate

Climate erosivity is portioned in the erosivity of the rainfall and runoff.

Rainfall erosivity is a function of rain intensity and kinetic energy. Rain intensity is related to two types of rain events, the short-lived intense storm where the infiltration capacity of the soil is exceeded, and the prolonged storms of low intensity that saturate the soil. Rainsplash erosion is closely related to rainfall through the detaching power of raindrops striking the soil surface. The ability of rain to cause rainsplash erosion is attributed to the mass, diameter and terminal velocity of the raindrops which affect the kinetic energy load of a rainstorm.

The runoff erosivity is dependent on the infiltrability of the soil and the effect of rain intensity through the contribution of rain to runoff. Runoff erosivity is a function of the shear stress of the flow exerted on the soil and the flow transport capacity. Soil detachment and rill initiation by surface runoff is related to flow depth or hydraulic radius and the shear velocity of the flow. The transport capacity is directly related to the amount and velocity of overland flow.

Soil

Soil erodibility affects erosion through its resistance to the impacting forces of rainfall and runoff, and through its influence on runoff initiation and the amount of runoff.

The role of soil texture is important in that large particles are resistant to transport because of the greater force required to entrain them and that fine particles are resistant to detachment because of their cohesiveness. The least resistant particles are silts and fine sands. The detachability of soil depends also on surface soil shear strength which is influenced by initial soil moisture, soil surface structure, structural stability and surface sealing, organic matter content, clay mineralogy, and chemical constituents. The soil's resistance to detachment and transport depends also on the amount of disturbance.

The amount of overland flow that is generated from a rainstorm is not only dependent on the rain amount and intensity, but also on the soil profile. The production of overland flow is affected by the surface depression storage, surface sealing, infiltration (during the rain and immediately after), and profile storage (adsorption on dry soil, macroporosity, permeability depth and antecedent soil moisture content).

Topography

The effects of topography on erosion are partitioned in the effects of slope steepness and slope length.

Erosion increases with increases in slope steepness as a result of respective increases in velocity and volume of surface runoff. Steeper slopes also show less infiltration and more runoff. The reason is that infiltration decreases significantly with increasing slope, due to the reduction in the time available for rainfall to infiltrate. Gentle slopes erode less, because there is more surface ponding and slower, deeper, overland flow which protects the surface against the impact of rain.

Slope length has an effect on the occurrence of the type of erosion. Soil creep due to rainsplash erosion close to the watershed gives way first to interrill erosion due to broad overland flow and then to erosion due to rill flow. The influence of slope length on erosion rates varies strongly depending on the presence or absence of rill erosion and the density of rills.

Vegetation

Vegetation affects erosion through its above-ground components, such as leaves and stems that absorb some of the energy of falling raindrops and running water, and the below-ground components, comprising the root system that contributes to the mechanical strength of the soil.

Vegetation, such as leaves and stems, absorb some of the energy of falling raindrops so that less is directed at the soil. The effectiveness of a plant cover in reducing erosion by raindrop impact depends upon the height and continuity of the canopy and the density of the ground cover. Increasing tree canopies will bring about greater rates of detachment unless the ground is protected by a litter layer. Whilst row crops greatly reduced the kinetic energy of the rainfall under the canopy, erosion was locally increased between the rows where the leaf drips can result in very high localized rainfall intensities which can considerably exceed infiltration capacities leading to the generation of runoff. Stemflow also concentrates rainfall at the ground surface.

Plant cover dissipates the energy of running water by imparting roughness to the flow, thereby reducing its velocity. The level of roughness with different forms of vegetation depends upon the morphology and the density of the plants, as well as their height in relation to the depth of flow. When the flow depth is shallow, as with overland flow, the vegetation stands relatively rigid and imparts a high degree of roughness. As flow depths increase, the grass stems begin to oscillate, disturbing the

flow and causing surface roughness to increase. With further increases in flow depth, the vegetation is submerged; the plants tend to lie down in the flow and offer little resistance, so that surface roughness decreases rapidly. Greatest reductions in velocity occur with dense, spatially uniform, vegetation covers. Clumpy, tussocky vegetation is less effective and may even lead to concentrations in flow with localized high velocities between the clumps.

The fine roots, 1-20 mm in diameter, interact with the soil to form a composite material in which root fibres of relatively high tensile strength reinforce a matrix of lower tensile strength. In addition, soil strength is increased by the adhesion of soil particles to the roots. Roots can make significant contributions to the cohesion of a soil, even at low root densities and in materials of low shear strength (Morgan, 1995; Lal and Elliot, 1994).

5. MITIGATION MEASURES

From the discussion of erosion processes (Chapter 4), it follows that the strategies for erosion control must be based on covering the soil to protect it from raindrop impact; increasing the infiltration capacity of the soil to reduce runoff; decreasing the erodibility of soil by improving aggregate stability; and increasing surface roughness to reduce the velocity of runoff. The various erosion control measures can be described under the headings of crop and vegetation management, soil management, mechanical methods and hydrological management (Table 5.1). For sediment-bound pollutants, management of sediment delivery to watercourses is critical.

Generally, preference is always given to crop and vegetation, and soil management measures. These are less expensive and deal directly with both the detachment (sediment production) and transport (sediment delivery) phases of erosion. They are also more easily fitted into an existing farming system. Mechanical measures are largely ineffective on their own since they are most often effective in controlling only the transport phase. Their main role is in supplementing crop and vegetation, and soil management measures, being used to control the flow of any excess water that arises. Many mechanical works are costly to install and maintain, and some create difficulties for farmers (Morgan, 1995).

5.1 CROP AND VEGETATION MANAGEMENT

Erosion control strategies of crop and vegetation management on cultivated land are aimed at establishing and maintaining good ground cover by using the protective effect of plant covers and by minimizing the period of bare ground. The feasibility of this is determined by how quickly, under the local climatic and soil conditions, the crops attain 40 to 50 % canopy cover, and by maintaining ground cover during periods with sparse canopy cover, for example, by leaving crop residue on the land after harvest and delaying ploughing to spring. Vegetative erosion control on grazing land is directed at maintaining a good vegetative cover by determining and maintaining suitable stocking rates, grazing on a rotational basis and controlled burning.

5.2 SOIL MANAGEMENT

Soil management is concerned with ways of preparing the soil to promote plant growth, and by improving its aggregate stability for increased infiltration and resistance to the shear stresses of raindrops and running water.

5.3 MECHANICAL METHODS

Mechanical structures are implemented to control the movement of water over the soil surface. A range of techniques is available and the decision as to which to adopt depends on whether the objective is to reduce the velocity of runoff, increase the surface water storage capacity or safely dispose of excess water (Morgan, 1995).

5.4 HYDROLOGICAL MANAGEMENT

Hydrological management is aimed at managing land in such a way as to prevent the spatial connectivity of runoff producing areas (Walter *et al.*, 2000; Endreny and Wood, 1999).

The AGIS and WOCAT web sites can be visited at www.agis.agric.za and www.wocat.org respectively for more information on erosion control measures.

Table 5.1 List of various erosion control measures (Morgan, 1995)

MITIGATION MEASURE	DESCRIPTION
CROP AND VEGETATION MANAGEMENT	
ROTATION	
Crop rotation	Growing of different crops consecutively in rotation so that a high rate of erosion under row crops is counteracted by low rates under other crop such as legumes and grasses
Grazing management	Moving of stock from one pasture to another in turn, to give time for the grass to recover
COVER CROPS	
STRIP-CROPPING	Growing of row crops and protection-effective crops in alternating strips aligned on the contour
RIPARIAN BUFFERS	Growing strips of grasses or woody species strategically placed in riparian areas and on footslopes
MULTIPLE CROPPING	
Sequential cropping	Growing of two or more crops a year in sequence
Intercropping	Growing of two or more crops on the same piece of land at the same time
MULCHING	
REVEGETATION	
Restoration of gullied lands	Planting of grasses, legumes, shrubs and/or trees area around gullies to increase infiltration and reduce surface runoff
Afforestation	Planting of trees
Restoration for pasture	Planting of grasses, legumes or shrubs
Embankment and cut slopes	Rapid establishment of a grass or legume cover on embankment and cut slopes to enhance slope stability
STREAMBANK PROTECTION	
AGROFORESTRY	
SOIL MANAGEMENT	
ORGANIC CONTENT	Addition of organic material as green manures, straw or as a manure
TILLAGE PRACTICES	
No tillage	Tillage is restricted to that necessary for planting the seed
Strip tillage	Preparation of the soil for planting along narrow strips with the intervening areas left undisturbed
Mulch tillage	Stubble-mulch tillage
Minimum tillage	Tillage practices using chiseling or disking to prepare the soil whilst retaining a 15 to 25 % residue cover
MECHANICAL METHODS	
CONTOURING	Carrying out ploughing, planting and cultivation on the contour
CONTOUR BUNDS	Earth banks, 1.5 to 2 m wide, thrown across the slope to act as a barrier to runoff
TERRACES	Earth embankments constructed across the slope to intercept surface runoff and convey it to a stable outlet

MITIGATION MEASURE	DESCRIPTION
WATERWAYS	The purpose of waterways is to convey runoff at a non-erosive velocity to a suitable disposal point
STABILISATION STRUCTURES	Play an important role in gully reclamation and gully erosion control
GEOTEXTILES	Is any permeable textile material used with foundation, soil, rock, earth or any geotechnical engineering related material
STREAM CHANNEL STABILISATION	
FLOOD CONTROL STRUCTURE	Construction of a flood relief (bypass) channel or culvert to accommodate flood flows
FENCING	Streambank and gully fencing to separate livestock from gullies and streams
HYDROLOGICAL MANAGEMENT	Creation of a mosaic pattern of areas with contrasting hydrological responses and raising the wetness threshold

6. ASSESSMENT OF PREDICTIVE ABILITY

6.1 INTRODUCTION

One of the key problems facing catchment and water resource managers is predicting fluxes of sediment in rivers. A number of models are being developed to better understand the complex erosion processes that occur in agricultural catchments, but are of necessity simplifications of reality (Hook, Fleming and Thomas, 1998). The variety of landscapes and climates within South Africa to which models need to be applied, the diversity of tasks and the variation in scales in both time and space to be modelled, require an overview of the diverse range of erosion models from which the modeller may choose the most appropriate. The following provides a general overview of the basic principles differentiating most models and their application.

6.2 DESCRIPTION OF MODELS

6.2.1 Types of Models

Erosion models may be classified according to a number of basic principles and assumptions, which influence the scale of analysis, the input data requirements, the type of output provided and the understanding of the underlying physical processes governing sediment yield source impacts. A primary distinction often made between different erosion models is whether they are deterministic or stochastic (Jewitt and Görgens, 2000). Stochastic models will not be further evaluated, because they are difficult to apply and the model user must have a thorough knowledge of statistical analyses (Hook *et al.*, 1998). All models considered in this review are deterministic.

Essentially, the differentiation between classes of deterministic models is one based on the level of complexity used to represent the real world process to be simulated. This ranges from qualitative rule-based models to empirical, conceptual and process models. Models can also be classified in terms of their spatial resolution and temporal scope. Spatially lumped models combine a number of possibly heterogeneous areas together and represent them as a single average. Spatially distributed models separate the area into homogeneous source areas and represent the relationships between them. Static models are time and state invariant while dynamic models can predict the change in a state such as soil water content, or a process such as runoff, sediment detachment and transport over time, taking into account time varying (known as time series) input data. Dynamic models may operate on time-steps ranging from seconds to months. Daily time-steps are often used because data collection on a daily basis is a common practice. Dynamic models may be either event-based, operating for the duration of a particular event such as an episode of rainfall and runoff, or they may be continuous and incorporate data collected continuously for months or years. By grouping and classifying models, we can evaluate aspects of their practical use without having to know each model in detail (Pegram and Görgens, 2001; Jewitt and Görgens, 2000; Hook *et al.*, 1998).

Qualitative rule-based models

The simplest form of a qualitative model is the "rule-based" model. Qualitative Rule-Based Models (QRBM) calculate or derive a value for an output variable given a set of input variables by using logical inference and makes use of "IF-THEN-ELSE" constructs. To predict changes from one state to another, simple IF-THEN type rules are applied in the form IF (some condition is met), THEN (the value of the state variable changes from one state to another). The rules forming the QRBM are often elicited from non-point source expert (specialist) opinions, the observations during site visits and catchment surveys, and experimentation (Jewitt and Görgens, 2000).

Empirical models

Empirical models describe erosion by some input on erosion factors and an output of soil loss or sediment yield, without using knowledge of the processes involved. Empirical models were necessary at a time when considerable plot and catchment data were available for analysis but the physical processes involved were not well understood (Onstad, Mutchler and Bowie, 1977).

Although good relations can be found between sediment yield and erosion factor(s), empirical models do not give insight into the erosion processes; are location specific and cannot be extrapolated to ungauged catchments with confidence; and lack the capacity to predict the effect of land management or climatic change (Hatton, Nicolli, Hairsine and Cresswell, 1998; Morgan, 1995).

Statistical equations

The first erosion models that were developed are those derived from regression analysis. These are usually simple statistical equations relating sediment yield to one or more catchment or climatic factors such as flow-duration, precipitation-temperature ratio, average catchment slope, % soil particles coarser than 1 mm in topsoil, and the aggregation index (Morgan, 1995; Onstad *et al.*, 1977).

Sediment delivery ratio

Sediment yield of a catchment can be predicted by multiplying the soil loss estimate (gross erosion which presents the total amount of eroded soil) with a sediment delivery ratio (SDR). The SDR represents the proportion of eroded soil that is delivered into the receiving surface water which accounts for the deposition of eroded soil during its movement from the source area. The soil loss is generally derived from an estimate of erosion from interrill and rill areas based on a soil loss equation, such as the Universal Soil Loss Equation (USLE) and the Soil Loss Estimator for Southern Africa (SLEMSA), corrected to take account of additional contributions for channel and gully erosion. These estimates of gross erosion, therefore, are open to considerable uncertainty (Pegram and Görgens, 2001; Walling, 1994).

There is no general equation for determining the sediment delivery ratio, but there are factors that provide guidance. However, considerable uncertainty surrounds these methods for calculating sediment delivery ratios. The failure to produce a generally applicable prediction equation for SDR is due partly to the complexity of sediment delivery processes and their interaction with catchment characteristics and partly to a lack of definitive assessments of the dependent variable itself (Walling, 1994).

Conceptual models

Conceptual models often have complex hydrological process simulations and an empirical sediment module (Pegram and Görgens, 2001). These models are based on spatially lumped forms of water and sediment continuity equations. The focus has been to predict sediment yields, primarily using the concept of the unit hydrograph (Nearing *et al.*, 1994).

Most of the conceptual models draw on the Modified Universal Soil Loss Equation (MUSLE; Williams and Brendt, 1977) approach. These models predict sediment yield by using the most important factors that affect sediment yield: hydrology, soil, slope length and steepness, vegetation, and erosion-control practices. The fundamental equation of MUSLE is:

$$Y_{sd} = \alpha_{sy}(Q_v \times q_p)^{2.47} K L S C P$$

where Y_{sd} = sediment yield from an individual storm in metric tons

Q_v = storm runoff volume in m^3

q_p = peak runoff rate in m^3/sec

K = USLE soil-erodibility factor

LS = USLE slope length and slope gradient factor
C = USLE crop management factor
P = USLE erosion-control practice factor
 α_{sy} , β_{sy} = model parameters

Storm runoff volume is predicted with a daily runoff model, and peak runoff rate is computed from storm hydrographs. Large catchments are divided into subcatchments, and sediment yield for a subcatchment is computed as for a small catchment. The MUSLE approach contains a built-in delivery ratio (Lorentz and Schulze, 1995)

Physically-based models

Physically-based models (also called process or mechanistic models) attempt to describe the physical processes and interactions governing sediment production and delivery. These models incorporate the continuity equation which is a statement of the conservation of matter as it moves through space over time. Almost all physically-based models are based on the scheme developed by Meyer and Wischmeier (1969), as shown in Figure 6.1.

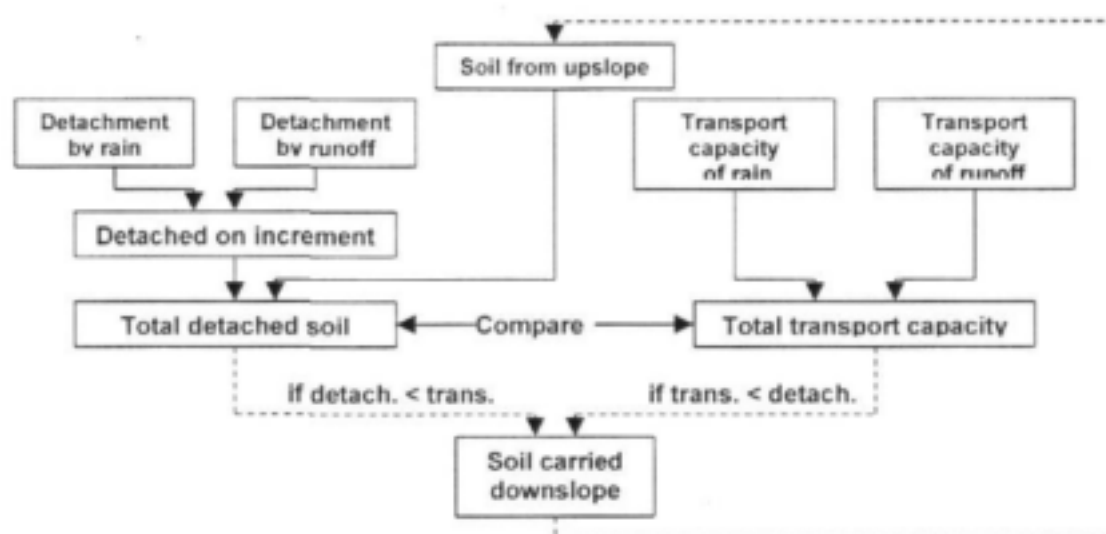


Figure 6.1 Scheme developed by Meyer and Wischmeier (1969) for the modelling of erosion processes.

Physically-based models are based on the fundamental concept that sediment load is assumed to be limited by either the sediment made available by detachment or by transport capacity. They simulate the sediment yield processes in a distributed way by using five main equations, namely particle detachment rate by raindrop impact, particle detachment rate by runoff, transport capacity by rainfall, transport capacity of runoff, and deposition rate in rill flows and channels. These models divide the catchment area into several segments. For each segment, the computed sediment load and transport capacity are compared. If the sediment load is less than the transport capacity, detachment occurs; but if the sediment load is larger than the transport capacity, deposition occurs. Physically-based models do account for spatial distribution of sediment yield (Morgan, 1995).

Since physically-based models simulate the erosion processes and the behaviour of sediment movement more accurately, they should provide more reliable results. They are theoretically the preferred models to be applied for predicting the impact of management alternatives, especially for ungauged catchments. However, physically-based models are the most expensive type of model to

apply due to the increased computer use and computation time, they contain numerous parameters that are generally difficult to measure or to estimate, they require large and detailed data sets, and often are difficult to use (Hatton *et al.*, 1998; Morgan, 1995).

6.2.2 Sediment Yield Models

Each model was developed with a different general purpose of understanding the sediment yield generated from catchments. When predicting sediment yield, decisions need to be made on the suitable level of realism (the extent to which actual physical components and processes of a system are represented), simplicity (data needed to run the model) and generality (the extent to which the model can be applied to different conditions) depending on the objectives of the modelling exercise (Hook *et al.*, 1998; Bingner, 1990). Table 6.1 summarises the models that were theoretically evaluated. An overview of the model components is discussed in the subsequent sections.

Model type

ACRU, ACTMO, AGNPS, EPIC, GAMES, HSPF, MUSLE and SWRRB are conceptual sediment yield models, while ANSWERS, CREAMS, EUROSEM, GUESS, Topog and WEPP are physically-based sediment yield models.

ACRU, AGNPS, ANSWERS, HSPF, MUSLE, SWAT, SWRRB were developed for small to medium sized catchments ($1-1000 \text{ km}^2$), and ACTMO, CREAMS, EPIC, EUROSEM, GAMES, GUESS, Topog and WEPP were developed for hillslope and field sized catchments ($<1 \text{ km}^2 / 100 \text{ ha}$).

Simulation of hydrology

The modified SCS curve number equation is used by ACRU, AGNPS, EPIC, MUSLE, SWAT and SWRRB to determine surface runoff. Daily runoff is calculated by the SCS curve number equation from daily rainfall and a retention parameter that is dependent on the soil moisture condition. Models such as AGNPS, GUESS, Topog and WEPP apply the Green-Ampt infiltration approach, and the kinematic wave approach to route surface runoff. Surface runoff can be estimated with CREAMS by the SCS curve number approach for daily rainfall or the Green-Ampt infiltration equation for hourly data. ANSWERS calculates runoff from the inflow and outflow rate of a cell from rainfall and the volume of water stored in a cell over a period of time. The calculated runoff is then combined with Manning's surface roughness equation for both overland and channel flow routing. The hydrology component of ACTMO portrays the catchment as a continuum of hydrological zones. The model simulates runoff as a cascade of flow over hydrological zones and through soil layers, both as downward seepage and as subsurface baseflow (Pegram and Gorgens, 2001; Hatton *et al.*, 1998; Morgan, 1995; Schulze, 1995; Bingner, 1990).

Table 6.1 List of models that were theoretically evaluated

Abbreviation	Name	Developed By	Aim	Time Step	Partition
ACRU	Agricultural Catchment Research Unit	Univ. of Natal – Department of Agricultural Engineering		Daily	Sub-catchment
ACTMO	Agricultural Chemical Transport	US Dept. Of Agriculture – Agricultural Research Service	Predict chemical movement from small catchments	Event Hourly	Hydrological zone
AGNPS	Agricultural NonPoint Source Pollution	US Dept. Of Agriculture – Agricultural Research Service	Provide estimates of runoff water quality from agricultural catchments	Daily Event	Cell
ANSWERS	Areal Nonpoint Source Watershed Environment Response Simulation	US Dept. Of Agriculture – Agricultural Research Service	Assess impacts of alternative management schemes on agricultural land	Breakpoint	Cell
CREAMS	Chemicals, Runoff and Erosion of Agricultural Management Systems	US Dept. Of Agriculture – Agricultural Research Service	Evaluate non-point source pollution from field-sized catchments	Daily Event, Breakpoint	Channel and hillslope
EPIC	Erosion Productivity Impact Calculator	US Dept. Of Agriculture – Agricultural Research Service	Effect of erosion/conservation on soil productivity	Daily Event	Field unit
EUROSEM	European Soil Erosion Model		Compute the sediment transport, erosion and deposition over the land surface throughout a storm	Event Breakpoint	Channel and hillslope
GAMES	Guelph model for evaluating the effects of Agricultural Management systems on Erosion and Sedimentation	Dickinson, Wall, Rudra,	Predict water erosion and the subsequent delivery of sediment from field to stream and downstream in agricultural areas	Daily	Field size land cells
GUESS	Griffith University Erosion Sedimentation System	Griffith University	Simulate the processes of erosion and deposition along a hillslope	Breakpoint	Hillslope
HSPF	Hydrologic Simulation Program – FORTRAN	US Environmental Protection Agency	Simulation of catchment hydrology and water quality	Daily	
MUSLE	Modified Universal Soil Loss Equation	Williams, Onstad & Foster	Predict daily, monthly, and annual sediment yield for basins	Daily	Sub-catchment
SWAT	Soil and Water Assessment Tool		Predict the effects of management decisions on water, sediment, nutrient, and pesticide yields on large and small river basins.		Sub-catchment
SWRRB	Simulator for Water Resources in Rural Basins	US Dept. Of Agriculture – Agricultural Research Service	Predict the effect of management decisions on water and sediment yields for ungauged rural basins	Daily Event	Sub-catchment
Topog model series		Commonwealth Scientific and Industrial Research Organisation		Daily Event	Channel and hillslope
WEPP	Water Erosion Prediction Project	US Dept. Of Agriculture – Agricultural Research Service	For use in soil and water conservation planning and assessment	Breakpoint Continuous	Channel and hillslope

A limitation in all the models, except ACTMO, HSPF and Topog, is that they do not consider shallow subsurface redistribution of water. Therefore, they cannot represent saturation excess flow generated by exfiltration of throughflow. This is expected to be a particular problem where these flows are a significant proportion of the total flow that affects erosion. Models such as Topog that simulate the expansion and contraction of saturation zones and exfiltration of throughflow already possess the basic structure to predict the areas of potential gully formation. The linking of these models with suitable routines that model gully initiation and development will give them greater versatility as predictive tools. EPIC takes account of a crack flow by taking account of the amount of cracking (Armstrong *et al.*, 1995; Smith, Goodrich and Quinton, 1995).

Simulation of erosion from overland flow

All the models, except EUROSEM, GUESS, HSPF and WEPP, make use of the USLE technology to various extents to predict sediment yield. Models like ACRU, ACTMO, AGNPS, EPIC, GAMES, MUSLE and SWRRB compute the sediment yield in a lumped way by using various modifications of the Universal Soil Loss Equation. Other models, like ANSWERS and CREAMS, make use of some of the USLE factors but maintain the structure of a physically-based model which incorporates the laws of conservation of mass and energy (Pegram and Gorgens, 2001; Morgan, 1995; Lorentz and Schulze, 1995; Bingner, 1990).

ANSWERS and CREAMS describe the interrill and rill detachment rate separately. They use a relationship that describes detachment of the soil particles by raindrop impact from the rainfall erosivity (CREAMS) or from the intensity of the breakpoint rainfall (ANSWERS) in combination with USLE's K, LS and C factors. Erosion from overland flow that causes rill erosion is calculated from the runoff volume and peak runoff rate (CREAMS) or from the flow rate (ANSWERS) in combination with USLE's K, LS and C factors. The transport capacity is based on Yalin's equation. By taking cognisance of both rainfall erosivity and runoff, sediment load is calculated for both detachment or transport limited conditions (Bingner, 1990).

GUESS provides a continuous analytical solution of erosion/deposition processes by relating sediment flux to its causative factors. In this model the amount of sediment transported off a field or a slope is determined by the amount of flow and the concentration of sediment. The erosion and deposition processes are described by rainfall detachment by splash, sediment deposition through settling and rate of sediment entrainment by overland flow in or between rills (Morgan, 1995).

EUROSEM calculates the local sediment concentration from the local erosion/deposition rate and local inflow of water containing sediments such as lateral inflows to a rill or channel. The term local erosion and deposition rate is composed of two independent erosion processes, rainsplash erosion, dependent on rain rate and water depth, and net hydraulic erosion caused by flowing water, or deposition from the surface water. Hydraulic erosion is treated as a dynamic difference between continuous processes of hydraulic soil particle detachment and deposition. EUROSEM use transport capacity relations describing a concentration, in terms of local hydraulic variables and particle size and density (Quinton, 1997; Smith *et al.*, 1995).

Development is underway to incorporate the WEPP erosion relationships in the AGNPS model. The erosion component of the WEPP model uses a steady-state sediment continuity equation, which calculates net values of detachment or deposition rates along the hillslope profile and in catchment channels. The erosion process is divided into channel, rill, and interrill components. Interrill erosion is calculated for rainsplash erosion whereafter the detached sediment is transported by shallow overland flow. Scour by rill and channel flow is calculated for the case when flow shear exceeds

critical shear of the soil and when sediment load is less than calculated sediment capacity (Hook et al., 1998; Morgan, 1995).

HSPF models sediment yield from pervious areas using Negev's model which simulates the generation and transport of fine soil particles by raindrop impact as a function of the land cover fraction during the growing season, a soil property factor and the precipitation during the storm period. These particles are available for transport if the surface runoff occurs during the storm period, otherwise they accumulate in the reservoir of fine particles which is available for subsequent periods. The delivery of particles is estimated as a function of these variables, a transport factor and a power factor (Pegram and Gorgens, 2001).

Simulation of channel erosion

Except for AGNPS, CREAMS, HSPF and WEPP, none of the other models have equations that specifically take account of channel bed erosion (Bingner, 1990). SWAT calculates the amount of channel erosion by the difference between transport capacity and sediment load (FitzHugh and Mackay, 2001).

Simulation of gully erosion

Most erosion prediction models emphasise the interrill and rill erosion aspects of the erosion cycle, but few models are capable of predicting ephemeral gully erosion. Of the evaluated models, only CREAMS, HSPF and WEPP claim to be capable of predicting ephemeral gully erosion.

None of the models, except Topog, have the capability to take account of the initiation and development of classical gullies. This is obviously a severe limitation if these models are to be applied to plan erosion control measures in catchments where gullies are the dominant source of sediment. Several studies have shown that the hydrographs predicted by sediment yield models, for relatively large events, showed good correspondence with measured hydrographs for both gullied and ungullied catchments, while sediment yields were only satisfactorily predicted where there were no gullies (Armstrong et al., 1995).

Simulation of sediment routing

The conceptual sediment yield models, ACRU, ACTMO, AGNPS, EPIC, GAMES, MUSLE and SWRRB that rely on the MUSLE technology, contain a built-in delivery ratio. The sediment routing technique of MUSLE is based on the assumption that sediment deposition depends upon settling velocities of the sediment particles, length of travel time, and the amount of sediment in suspension. The physically-based models, ANSWERS, CREAMS, EUROSEM, GUESS and WEPP, route the sediment from interrill and rill areas on a hillslope to the channel waterways through the continuity equation of the conservation of matter as it moves through space over time. Interrill areas act as sediment feeds to the rills, or small channel flows. The rills and small channels on the hillslopes feed sediment to catchment channels. However, for larger catchments, a sediment routing technique is needed to route the sediment yield from subcatchments through streams and valleys to the outlet of the catchment (Lorentz and Schulze, 1995; Bingner, 1990).

SWRRB routes sediment from subcatchments through ponds, reservoirs, channels and a flood plain. Sediment is routed through ponds and reservoirs by simulating the inflow and outflow sediment concentration (sediment yield/volume water) of the ponds and reservoirs. The channels and flood plain routing model is composed of the deposition and degradation equations operating simultaneously. ANSWERS and AGNPS route the detached sediment from raster cell to cell into channels, which are then routed downstream to the catchment outlet. The basic routing equation is derived from the steady-state continuity equation (Bingner, 1990).

Modelling units

The method used by a model to describe the non-homogeneity of a field, whether by space or time, is important. How a model describes the variation in a catchment can determine how well a model can predict the runoff and sediment yield. Too much or too little description of a catchment can have detrimental effects on the application of a model. If a model requires a user to input long and complicated sets of data for a catchment then that model may not be accepted, except by researchers. On the other hand, if a model requires little input by the user, then the model becomes site-specific or produces inaccurate results (Bingner, 1990).

Spatially lumped models, like ACRU, ACTMO, EPIC, MUSLE, SWAT and SWRRB, assume that a field/sub-catchment that is simulated acts homogeneously throughout, although SWRRB does allow multiple fields to be simulated within a catchment. ACTMO portrays the catchment as a continuum of hydrological zones.

CREAMS, EUROSEM, GUESS, Topog and WEPP apply the technique of segmenting the catchment's channels and overland flow paths. Each segment can thus have a different set of input parameters to describe that part of a catchment. The description of the catchment is thus limited to an average flow path. With a complex field of variable slopes and different types of soils, this description may become difficult to define. These models are called spatially semi-distributed models.

Grid-based models that are spatially distributed models, such as AGNPS, ANSWERS and GAMES, put much of their programming into describing a catchment through the use of cells, grids or elements. Each cell describes the particular characteristics of the catchment required for the equations used at that point. Runoff and sediment can then be routed through these cells, thus each cell is affected by the upstream cells. These models require extensive input concerning square cells throughout the catchment. This may better approximate the flow of the runoff based on the slope, direction of flow, and field conditions within each cell. Using cells requires the user to have a detailed knowledge of the catchment studied. For large catchments, the input parameters required by the models may require more time to develop than most users are willing to allow. As technology advances, especially in GIS, a detailed knowledge of a catchment may be an item that computers can easily provide (Hook *et al.*, 1998; Schulze, Angus, Lynch and Smithers, 1995; Bingner, 1990).

Simulation of other non-point source pollutants

Adsorbed and dissolved phosphorus concentrations in the top 10 mm of soil are modelled in ACRU using an adsorption process. During a rainfall event, the dissolved and adsorbed fractions are recalculated and the resulting concentrations are applied to the surface runoff and sediment yield to estimate phosphorus washoff. A similar approach was developed for *E.coli* loading (Pegram and Gorgens, 2001; Kienzie, Lorentz and Schulze, 1997).

ACTMO predicts the concentration of a chemical in runoff, the total amount transported by runoff and sediment, as well as the concentration and location of chemicals remaining in the field. Nutrients are modelled in ANSWERS using correlation relationships between chemical concentrations and sediment yield and runoff. A submodel of EPIC for the N and P budget includes fertilization, organic and inorganic transformations, crop demand and uptake of N and P. It includes denitrification, leaching of nitrate, and losses by wind and water erosion. SWAT predicts the effects of management decisions on nutrient, and pesticide yields on large and small river catchments (Pegram and Gorgens, 2001; Wang, Buland, Dayton and Dihuzio, 2000; Hook *et al.*, 1998).

CREAMS predicts the loss of dissolved and adsorbed chemicals, both nutrients and pesticides. Dissolved and adsorbed nutrient and pesticide concentrations are associated with surface and

sediment yield. The partitioning of these contaminants between the soluble and solid phase is performed using a simple isotherm model. The SWRRB and AGNPS models have been developed based on the CREAMS modelling routines. The AGNPS model predicts nitrogen, phosphorus and chemical oxygen demand concentrations in the runoff and the sediment, for single storm events for all points in the catchment. In addition it considers nutrients and chemical oxygen demand from animal feedlots (Pegram and Görgens, 2001; Hook *et al.*, 1998).

Two alternative approaches can be employed in HSPF to estimate contaminant loads; either via potency factors or via detailed adsorption-desorption algorithms. HSPF-ARM performs detailed simulation of contaminant production, delivery and transport, particularly for nutrient and pesticide washoff from agricultural catchments. It uses an adsorption-desorption equation to simulate the transport and partition of contaminants, and represents contaminant transformation during delivery and transport (Pegram and Görgens, 2001).

Gully erosion models

Empirical models

Several (statistical) empirical models for describing and predicting gully growth have proved useful for the working conditions where they were formulated. The resulting equations typically link headcut retreat with parameters such as drainage area above the gully head, rainfall depth, erodibility, height of the headcut, relief energy of drainage basin and runoff response of the drainage area. Their in-built regional bias, however, may hamper extrapolation (Bocco, 1991; Poesen, Nachtergaele, Verstraeten and Valentin, 2003).

Ephemeral gully erosion models

At present, only a few models claim to be capable of predicting ephemeral gully erosion rates, i.e. CREAMS, GLEAMS (Groundwater Loading Effects of Agricultural Management Systems; Knisel, 1993), EGEM (Ephemeral Gully Erosion Model; Merkel, Woodward and Clarke, 1988, Woodward, 1999) and WEPP. The channel erosion routines from both GLEAMS and WEPP are slightly modified procedures from the CREAMS channel erosion routines. In these models, concentrated flow detachment rate is proportional to the difference between (1) flow shear stress exerted on the bed material and the critical shear stress, and (2) the transport capacity of the flow and the sediment load. Net detachment occurs when flow shear stress exceeds the critical shear stress of the soil or gully bed material and when sediment load is less than transport capacity. Net deposition occurs when sediment load is greater than transport capacity. Although these models claim to have a great potential in predicting sediment production by ephemeral gully erosion, they have never been thoroughly tested for this erosion process. Furthermore, these models lack routines to predict the location of gullies (Poesen *et al.*, 2003).

An approach to predict locations in the landscape where gully heads might develop is represented by the threshold concept. This concept is based on the assumption that in a landscape with a given climate and land use there exists for a given slope a critical drainage area necessary to cause gully incision. The threshold curves for ephemeral gullies reflect a critical flow intensity which needs to be exceeded in order to initiate gully. Once the ephemeral gully thresholds are established, one could predict the effect of land use on the gully network density and location using models that were developed to assess the average shear stress exerted by the flow on the gullies. Quantification of the erosive power of overland flow is possible through the use of various flow intensity measures such as shear stress, shear velocity or stream power. Such predictions can be made using routing algorithms describing how flow and water-borne material will be routed over Digital Elevation Models. These

algorithms allow one to calculate drainage area and slope for each point in a catchment (Bocco, 1991).

The downslope end of ephemeral gullies are predicted where massive sediment deposition and fan building occurs where either surface roughness increases suddenly (e.g. different land use) or where local slope gradient decreases. Consequently, ephemeral gully length can be derived by routing concentrated flow from the gully head towards the fan at the gully end (Poesen *et al.*, 2003).

Classical gully erosion models

Compared to interrill, rill and ephemeral gullies, little is known about the processes and factors controlling classical gully erosion. The eroded sediment volumes by classical gully erosion were not significantly related to a measure of overland flow intensity and the location of classical gullies was only in a limited number of cases related to topographically-controlled concentrated overland flow crossing a bank. In most cases, the location of the classical gullies along existing banks could not easily be explained and depended often on the presence of macropores (biopores and tension cracks) and the occurrence of mass movement processes (slumping and soil fall) (Bocco, 1991).

There have been recent approaches to modelling headcut formation, headcut geometry, and headcut migration. Gully formation is very intensive during the period of initiation, when morphological characteristics are far from stable. This period is relatively short, about 5% of a gully's lifetime. For the most part of a gully's lifetime, its size is nearly stable. These two stages of gully development led to two types of gully erosion models: (1) dynamic models to predict rapid changes of gully morphology at the first period of gully development; and (2) static models to calculate final morphometric parameters of stable gullies (Sidorchuk and De Roo, 1999).

Dynamic models

Sidorchuk (1998) presented a dynamic model for the first stage of gully development. The dynamic gully model is based on the solution of the equations of mass conservation and gully bed and wall deformation. The model of straight slope stability was used for predicting gully side wall inclination. The rate of gully erosion is controlled by water flow velocity, depth and turbulence, as well as soil texture, mechanical pattern and protection by vegetation. These characteristics are combined in equations of mass conservation and deformation. The equation of mass conservation defines the sediment budget in the channel reach, the sediment storage in the channel flow, the sediment flux from the banks and from the downward flux. The equation of deformation defines the change of gully bottom elevation and bank co-ordinates according to the sediment budget. This dynamic model that has been developed for Australian conditions is included in the Topog suite of models.

Hanson, Robinson and Cook (2001) have developed a deterministic physically-based model to predict headcut advance that assumed headcut migration by undercutting and vertical headwall failure. The model uses a nappe profile component, a boundary stress component, an excess stress erosion rate component, and a mass-wasting component. No other contributing factors such as seepage or weathering are included in the model. This model predicts headcut failure by erosion of the headcut face and the downstream channel due to hydraulic stresses in the plunge pool region of the headcut. The model includes a soil strength component for prediction of headcut failure and underlying assumptions related to debris clean out downstream of the headcut. The rate of headcut migration is based on measurable input parameters of soil strength, soil unit weight, erodibility and critical shear stress, initial overfall height, depth to a more erosion-resistant layer, backwater level, and discharge.

Static models

The static gully model of Sidorchuk (1998) is based on the assumption of final morphological equilibrium of a gully; when averaged for several years, elevations and the width of gully bottom do not change. This stability is associated with a negligible rate both of erosion and sedimentation at the gully bottom. It means that flow velocity is less than the threshold value for erosion initiation, but is more than the critical velocity of wash load sedimentation. This static model was validated and verified in the Mbuluzi River catchment of Swaziland that has similar environmental conditions to Australia where it was developed (Sidorchuk, Marker, Moretti and Rodolfi, 2003).

6.3 SCALE AND EROSION MODELLING

6.3.1 Operational Scale

The operational scale is the scale at which erosion studies such as modelling exercises, and management action focus. It effectively includes the integrative modelling scale and, ideally, should be compatible with the scale at which managers focus their operations. However, there is unlikely to be any single preferred operational scale (Jewitt and Görgens, 2000).

The starting point for all modelling must be a clear statement of the objective of the intended study. When predicting erosion, decisions need to be made on whether the prediction should be for a year, a day, a storm or for short time periods within a storm; and whether it should be for a field, a hillslope, small catchment, large catchment or a drainage basin. These differences in temporal and spatial perspectives will influence the processes which need to be included in the model, the way they are described and the type of data required for model validation and operation.

The objective of the study also has implications for how well a model must perform. Screening models are simple in concept and designed to identify problem areas. Usually it is sufficient if predictions are of the right order of magnitude. Assessment models need to predict with greater accuracy because they are applied for evaluating the severity of erosion under different management systems and may be intended as design tools for the selection of conservation practices.

The scale of operation must also be considered. The detailed requirements for modelling erosion over a large catchment differ from those demanded by models for small catchments or from a short length of hillslope. Scale influences the number of factors which need to be incorporated in the model, those that can be held constant and those that can be designated primary factors around which the model must be constructed (Morgan, 1995).

6.3.2 Elements of Non-point Source Assessment Associated with Sediments

A conceptual framework on the four elements of the non-point source impacts of erosion on water quality provides a platform for the approaches for assessment and management of erosion in a catchment. The last element, water use, will not be discussed, since it is not in the scope of this study.

Production

The sources of sediment production include the detachment of soil particles and re-entrainment of deposited sediment at the interrill and rill areas (see Section 4.2.3), and in gullies (see Section 4.2.2) and stream channels (see Section 4.2.1). The production element falls in the source zone of the fluvial classification system of Schumm (1977). Management of the production element follows a source-directed philosophy which is mainly directed in agriculture towards crop and vegetation management (see Section 5.1), and soil management (see Section 5.2).

Delivery

The delivery of sediment to rivers includes the transport of detached soil particles and re-entrained sediment from interrill and rill areas, gullies and stream channels to small (ephemeral) streams (see Section 4.2.2), and then to rivers (see Section 4.2.1). The delivery element falls in the source zone of Schumm's fluvial classification system. Management of the delivery element follows a source-directed philosophy which is mainly directed in agriculture towards mechanical methods (see Section 5.3), riparian buffer zones, vegetative filter strips (see Section 5.1), and hydrological management (see Section 5.4).

Transport

The transport element refers to the movement of sediment in the rivers, impoundment lakes and estuaries (see Section 4.2.1). The transport element falls mainly in the transfer zone and also in the deposition zone of Schumm's fluvial classification system. Management of the transport element focuses on the fate of sediment once already in the aquatic environment, and is therefore not consistent with source-directed management philosophy. The management of the transport component is beyond the scope of this study (Pegram and Görgens, 2001; Wasson, 1998).

The above conceptual framework enables analysts to identify particular element(s) they are addressing and thus what contribution their analysis is making to the understanding of the catchment for management purposes. This is particularly important as the location of most water quality monitoring is in the transport element, representing the combined impact of production, delivery and transport. This complicates the assessment of contributions from particular sources to monitored sediment load. In order to disentangle the production, delivery and transport elements, Pegram, Görgens and Otterman (1997) have adopted the following definitions:

Potential

The sediment potential is the quantity of sediment available at the source for delivery, and represents the production element. The sediment potential can be predicted as the soil loss from interrill and rill areas, gullies and stream banks.

Yield

The yield is the quantity of sediment being delivered from the source areas to the rivers, representing the combined effect of production and delivery. The yield can be predicted as the sediment yield from hillslopes and for small catchments ($1-10 \text{ km}^2$).

Export

Export is the quantity of sediment leaving a catchment, which represents the combination of the production, delivery and transport. The export can be predicted as the sediment yield with models which have a good instream sediment routing method for medium-sized catchments ($10-1000 \text{ km}^2$), and with higher uncertainty for large catchments ($>1000 \text{ km}^2$) with a larger sediment storage (Wasson, 1998; Pegram *et al.*, 1997).

6.3.3 Levels of Non-point Assessment and Erosion Modelling

The water quality assessment represents a process, beginning with a scoping of the water quality issues and leading towards the selection of *inter alia* non-point source management measures. A hierarchical approach is followed, with each level of assessment building on information provided by the previous investigation. Assessment techniques generally get more detailed with progression through the levels of assessment and the assessment techniques require greater resolution and predictive powers for the latter levels. The "Guide to non-point source assessment", developed by

Pegram and Görgens (2001), can be consulted for detailed descriptions on non-point source assessment approaches, procedures and techniques. Only a broad overview of non-point source assessment techniques specifically related to sediments and erosion prediction will be presented in the following sections and are summarised in Table 6.2.

Knowledge-based approaches

Knowledge-based approaches provide qualitative investigations, based on a combination of expert (specialist) opinions, catchment surveys, and consultation for anecdotal evidence of local inhabitants (Pegram and Görgens, 2001). Knowledge-based approaches are not discussed since they are beyond the scope of the project.

Data analysis techniques

Data analysis techniques depend upon the statistical and stochastic analysis of monitored water quality and/or hydrological data. The ability to differentiate between production, delivery and transport, as well as different non-point source areas, is limited, even in the most highly monitored systems (Pegram and Görgens, 2001). The simple statistical relationships developed between sediment yield and one or more catchment or climatic factors from regression analyses have a limited predictive power outside the observed conditions and lack the capacity to predict the effect of land management change (Section 6.2.1.2). Data infilling and stochastic techniques are not discussed since they are beyond the scope of the project.

Potential and hazard maps

Potential mapping

Potential mapping is based on an assessment of the quantity of eroded sediment available at the source for delivery (also called soil loss) for the different source areas throughout a catchment (Pegram *et al.*, 1997). The sediment potential can be predicted for interrill and rill areas by using the Universal Soil Loss Equation (USLE) or the Soil Loss Estimator of Southern Africa (SLEMSA) (Morgan, 1995). However, soil loss estimates from USLE or SLEMSA can under-estimate the quantity of sediment produced in severely gullied catchments since they do not take account of the sediment produced from gullies or stream channels (Wasson, 1998). Potential maps only provide an indication of the potential erosion impact, not the actual sediment yield for a source area or export from a catchment (Pegram and Görgens, 2001). Potential maps identify land uses and areas requiring further detailed non-point source analysis and focused management attention, in terms of land use planning and strategy intervention (Kienzie *et al.*, 1997).

Hazard mapping

The development of hazard maps extends the potential maps by incorporating sediment delivery into the estimation procedure. This provides a spatial indication of the relative impact of different non-point sources. Hazard maps are based on the use of GIS coverages which reflect the factors governing the movement of sediment from the source into the receiving surface waters, usually combined with the relevant potential maps. The representation of these delivery factors may be empirically derived (Section 6.3.2.5) or model simulated (Section 6.3.2.6) (Pegram and Görgens, 2001). However, almost all of the assessment techniques used to determine hazard maps do not take account of sediment delivery from active gullies and sediment delivery can therefore be under-estimated for gullied catchments (Wasson, 1998).

Gully density mapping

In gullied catchments, potential and hazard mapping should be complemented with a gully density map. In gullied catchments, mean annual sediment yield is most often a linear function of gully density, which accounts for most of the variation in sediment yield. Gully density maps are produced by plotting the gullies from aerial photomaps and then generalising the information to produce a map of gully density (Wasson, 1998; Morgan, 1995).

Table 6.2 The objective, management decision and assessment techniques at different levels of assessment (Pegram and Görgens, 2001; Pegram *et al.*, 1997)

Level Of Assessment	Objective	Management Decision	Assessment Technique*
Scoping	- Identify the critical water quality concerns - Whether non-point sources contribute significantly to water quality problems - Which sub-catchments require further NPS assessment	The water quality issues to manage	- Knowledge-based approaches - Data analysis techniques - Potential mapping - Unit area loading
Evaluation	- Identify and describe key areas and activities of concern	Key areas, activities and processes of concern	- Data analysis techniques - Potential mapping - Unit area loading - Loading functions - Simple process models - Detailed process models - Single source models
Prioritisation	- Prioritise sources for management - Describe main processes from priority sources - Evaluate manageability of a source and opportunities and constraints for management	Priority sources and management strategies	- Knowledge-based approaches - Loading functions - Simple process models - Detailed process models - Single source models - Heuristics
Selection	- Evaluate NPS impact of management actions - Evaluate social, economic and institutional viability - Select best management practice	Appropriate actions to achieve the specified strategies	- Simple process models - Detailed process models - Single source models - Heuristics
Operation	- Identify and select actions during implementation	Ongoing operational decisions	Beyond scope of project
Auditing	- Monitor and evaluate the results	Reassessment, replanning or further implementation	Beyond scope of project

* The specific application of an assessment technique will vary according to the level of non-point source assessment. Consult the "Guide to non-point source assessment" by Pegram and Görgens (2001) for discussions on the specific application of an assessment technique at an assessment level.

Unit area loading

Unit area loads (often referred to as export coefficients) are empirical estimates of the sediment load for a particular land use. However, great care should be taken when applying these coefficients for agricultural land uses, since the variation in sediment loads within a land use category is often greater than between categories due to the significant impact of natural characteristics (e.g. soils, topography and climate) and agricultural management practices (Pegram and Görgens, 2001). The limitations of empirical estimates in extrapolating data to ungauged catchments and predictive capacity of the effect of land management are discussed in Section 6.2.1.2.

Loading functions

Loading functions use simple sediment yield estimation techniques to describe sediment fluxes. Sediment yield of a catchment is estimated by multiplying the predicted soil loss (quantity of eroded sediment) with a sediment delivery ratio (SDR) to account for the deposition of detached soil particles during their movement from the source area. Soil loss is most often predicted with USLE or SLEMSA. The annual sediment yield from a source area is estimated as the product of the sub-catchment sediment delivery ratio, the area and the annual soil loss (Pegram and Görgens, 2001; Morgan, 1995).

It must be kept in mind that USLE and SLEMSA predict soil loss for interrill and rill areas and do not take account of the sediment originating from gullies. The occurrence of active gullies in a catchment often dominates the sources of sediment production in a catchment. Not only are gullies important sources of sediment, but, because the sediment production sources and mode of transport are adjacent to the watercourses, the SDR is higher than for ungullied or lightly gullied catchments (Wasson, 1998).

Considerable uncertainty surrounds the methods for calculating sediment delivery ratios. The failure to produce a generally applicable prediction equation for SDR is due to the complexity of sediment delivery processes and their interaction with catchment characteristics (Section 6.2.1.2). It should also be noted that sediment yield estimates include suspended solids, settleable solids and bedload, and that a large portion of the total annual sediment load occurs during a couple of storm events, which are often missed by routine grab-samples. Therefore, loads estimates from observed suspended solids data tend to be only 20% to 35% of the sediment yield, because this is only a portion of the sediment load and there is an under-measurement bias in the grab-samples.

The sediment-bound pollutant loading function is calculated from the product of the enrichment ratio, the mass concentration of the soil particle adsorbed pollutant associated with the source area and the sediment yield (Pegram and Görgens, 2001).

Simple process models

Simple process models use simplified deterministic representations of the processes governing sediment production and delivery. They simplify the representation of interactions between hydrology and sediment yield, but require the development of computer code to represent the more refined process, temporal and/or spatial detail. These models usually consist of separate hydrological and sediment modules that are simulated in a sequential manner, starting with the hydrology and followed by the sediment transport. However, this increased complexity generally implies greater data requirements (Pegram and Görgens, 2001).

Simplified process models often have fairly complex hydrological process simulations, but are described with this group if they simplify the sediment processes. These models are classed as conceptual erosion models (Nearing *et al.*, 1994) and are discussed in Section 6.2.1.3. An evaluation of several conceptual models (also called simple process models by Pegram and Görgens (2001)) is discussed in Section 6.2.2.

Detailed process models

The detailed process models incorporate detailed contaminant processes into the simulation of contaminant movement and transformation in soil and water, and do not use potency factors as the simplified models (Pegram and Görgens, 2001). In terms of the sediment modules, most of the detailed process models can be classed as conceptual erosion models discussed in Section 6.2.1.3.

Single source models

Most of the non-point source assessment techniques discussed in Sections 6.3.2.1 to 6.3.2.6 have been oriented towards analysis of multiple source areas at a catchment scale and will fall into the category of screening erosion models according to Morgan (1995). Single source models represent the processes and activities governing production and delivery of sediment from a single source area, with the aim of evaluating the effectiveness of alternative management interventions in order to selection of best management practices for that source area (Pegram and Görgens, 2001). These models, which fall into the category of assessment erosion models, need to predict with greater accuracy because they are applied for evaluating the severity of erosion under different management systems and may be intended as design tools for the selection of conservation practices (Morgan, 1995). Almost all of the single source models are classed as physically-based models (Nearing *et al.*, 1994) and are discussed in Section 6.2.1.4. An evaluation of several physically-based models is discussed in Section 6.2.2.

The biggest disadvantage of single source and detailed process models are that they tend to be very data intensive and as such are limited to areas for which there has been intensive data collection (Pegram and Görgens, 2001; Morgan, 1995).

Heuristics

Heuristics are rule-based or expert systems that identify the main processes and activities governing the production and delivery of sediment from source areas and indicate the effectiveness of alternative management practices. In other words, they are the qualitative equivalent to the single source models presented in Section 6.3.2.7. Heuristics are classed as qualitative rule-based models discussed in Section 6.2.1.1.

Heuristics are typically developed through empirical observation and experience with actual physical systems or research prototypes. As such, they are generally based on reliable data, but unfortunately these results are often contradictory, due to site-specific variations. The skill in using heuristics is to know which results are most applicable in a particular context. Thus, as with most non-point source assessment techniques, the application of heuristics requires experienced analysts with an understanding of both the literature values and the site-specific conditions under which they must be applied (Pegram and Görgens, 2001).

6.4 MODEL INTEGRATION

The term model integration is used as a generic term for linking models, i.e. exchange or transfer of input and output between models, by both simple and complex means (Jewitt and Görgens, 2000). There are a number of scaling problems in the context of erosion model integration which relates to the dominant processes and factors at the various scales, model conceptualisation and to keep within reasonable computational limits. Extending the spatial and temporal scales of soil erosion models based on the simulation of hydrological events needs to address interactions and feedback that become important at a coarser scale (Kirkby *et al.*, 1996).

6.4.1 Types of Model Integration

Linking models in series

Series type integration is most common and involves taking the output from one model and using it as input for the next. This implies all calculations of one model need to be finished before another model can start its work. This type of integration does not result in the formation of a new model, rather it leaves the models as they were and usually involves directing the output from one model to a storage system which enables it to be used as input for the next model, once the first model has completed its run. Converting the output of the first model from its unique format to the unique input format of the other model involves the use of some sort of data transformation program. This is a fairly common approach, since it requires little if any model software modifications.

Series links are characterised by a "poor" or "loose" link and "low usability". Costs of implementation, however, are usually low. This type of link does not allow feedback between the two systems. However, communication through the input/output files makes this type of loosely coupled models very flexible (Jewitt and Görgens, 2000).

Linking models in parallel

Parallel linking of models or model components involves using the output from one component as input to the next, in the same time step, and thus has the potential to accommodate feedback between the models. This is also referred to as "deep" or "definitional" integration. Deep integration provides a common interface and transparent file or information sharing and transfer between the respective components and may yield a new model which combines two (or more) components. The available literature reports three methods of linking models in parallel.

The first involves the restructuring of the models, so that they read the same database, but more significantly, so that they can be compiled into one executable program. This method is most commonly referred to as an "embedded system configuration". One component, the master, has the ability to invoke actions by the agent components. Such embedded systems provide a higher degree of efficiency in linking of capabilities and higher usability than simple series-based linking.

A second method of linking models in parallel is one which is currently receiving much attention worldwide. Federated database research involves the coupling of multiple databases maintained by autonomous, heterogeneous database management systems. Thus, a programme, separate from the models to be linked, may operate these models and control their input and output functions in such a way that a parallel link is achieved. Control of the linking processes rests with a message passing interface whilst the different components retain their own database access functions and remain as separate standalone executables. In other words, data input, output and transformation processes are retained by the individual components.

Closely related to the use of federated databases is the use of distributed database management systems, the third method of linking models in parallel. A distributed database management system (DDBMS) is a single logical database that is physically distributed on several computers, communicating through a computer network, usually a Local Area Network (LAN). It provides multiple access to the database and mechanisms to prevent conflict in update, retrieval and backup of data. DDBMS can be very useful for very large multi-component modelling systems with large databases where most users are concerned with only a small subset of the database, while still permitting access to the user with broader needs (Jewitt and Görgens, 2000).

6.4.2 Approaches of Model Integration

Universal model

Should we resolve to model integration, the option of constructing a 'universal' model will inevitably suggest itself and deserves to be considered. The attractions of such a synthetic model are obvious, especially if it includes data standards and operates across the diverse computing platforms. The cautions against such a complex model need equally serious consideration. Peer review of such a proposal is highly desirable and we should be aware of the philosophical and practical pitfalls.

Example: HSPF comprehensive model package

The Hydrological Simulation Program Fortran (HSPF) model is a comprehensive package consisting of a suite of computer routines that can simulate the hydrologic and associated water quality processes on and under the land surface and in streams and well mixed impoundments in urban and/or rural catchments. HSPF is one of the few models which combines both the generation and transport of pollutants across the land surface to watercourses and the transport and fate of pollutants in streams, rivers and reservoirs.

The model's major potential strength is its ability to simulate water quality components both on the land surfaces and in the river channel. It is a comprehensive model of catchment hydrology and water quality that allows the integrated simulation of land and soil contaminant runoff processes with in-stream hydraulic and sediment-chemical interactions. The result of this simulation is a time history of the runoff flow rate, sediment load, and nutrient and pesticide concentrations, along with a time history of water quantity and quality at any point in a catchment. HSPF models sediment yield from pervious areas and for impervious surfaces. The model simulates transport of sand, silt and clay and the re-suspension and settling thereof.

The model is built on a systematic framework in which a variety of process modules permit the continuous simulation of a comprehensive range of hydrologic and water quality processes. The framework evolved from a top down approach emphasising structured design and the model is planned around a time series management system. HSPF incorporates catchment-scale the Agricultural Runoff Management Model (ARM) and the Non-Point Source (NPS) models into a basin-scale analysis framework.

The drawback to HSPF is the data intensive nature of the input parameters required. HSPF is a comprehensive model and has extensive data needs, both for input as well as for calibration. Many of the parameters are not physically-based, and thus the model needs to be calibrated with local data, which limits its use in data poor situations (Jewitt and Gorgens, 2000; Van Rensburg and Dent, 1997).

Tightly-coupled modelling suites

Perhaps a more achievable and efficient way forward would be to develop a model manager that allows existing models to be interconnected and substituted. This framework would ideally operate at the model subcomponent or algorithm level (linking models in parallel) and would allow modelling to be customised for particular needs through selection of an appropriate suite of submodels. The extension of this framework to cover a diversity of erosion models at the algorithm level would be a daunting and expensive task, and could be limited by the availability of source code and by commercial considerations.

Example: MEDALUS-MEDRUSH modelling suite

The MEDRUSH model is developed from innovative sub-catchment models nested within a catchment routing framework and assembled to form a large scale model. The MEDRUSH model is embedded in a GIS system which analyses digital elevation models (DEMs), remotely sensed vegetation/land cover maps and thematic soil and geology maps to subdivide the area into sub-catchments of 5-10 km², and

provide soil, surface and vegetation cover parameters within each. Each sub-catchment has been broken down into a series of hillslopes (also called flow strips), and from the ensemble of hillslopes for a given catchment, a "representative hillslope" has been defined. For the representative hillslope, flows of water and sediment have been combined within a linearised routing model to give totals for the whole catchment, and also for its major tributaries, which may be compared with hydrological data. This strategy provides comparability with the MEDALUS hillslope catena model and detailed predictions of areas liable to degradation, while at the same time allowing integration of water and sediment yields up to the large catchment scale (Geeson and Thornes, 1996).

The hillslope catena MEDALUS model represents four interacting sub-models for the atmosphere, vegetation, soil and surface systems. These four sub-systems are defined at each of a series of points down a hillslope catena or flow strip, which are connected by kinematic routing of overland flow, where appropriate subsurface flow, and grain-size selective sediment transport. Simulation of vegetation growth gradually modifies soil organic matter.

The MEDALUS model is developed for a single hillslope over a period of up to 10 years, whereas in MEDRUSH up to 200 simplified hillslopes are included within a catchment of up to 2000 km², described by a DEM and a series of thematic GIS layers, and the model is intended to run for up to 100 years. In this way the simplified hillslope catena models can be used to predict catchment change across areas which are large enough to have significance for catchment management planners. Simulation of the hydrological and erosion processes, are therefore handled somewhat differently in the MEDALUS and MEDRUSH models. For MEDALUS, the processes that control the spatial distribution of infiltration, overland flow and sediment transport are simulated in detail by using 5-15 minute interval data to simulate conditions using a variable time step. In the MEDRUSH model, the hydrological routines are inevitably somewhat simpler and the model runs in basic time steps of one hour. Similarly, the flow is statistically distributed across a rough topography (Kirkby *et al.*, 1996).

Loosely-coupled modelling suites

An alternative way to either one single 'super-model' or a modelling framework which allows models to be linked in parallel, is a larger catchment decision support system from which the outputs from several strategic, issue-specific submodels can be used without the models necessarily being formally connected (linking models in series). There will also be a role for models and other computer programs which summarise outputs of 'research models' and make it more available to client groups. It will also be possible to identify qualitative associations between processes and landscape characteristics, as well as by quantifying causal relationships by screening type of models before more intensive predictions are made (Hatton *et al.*, 1998).

Example: Namoi basin modelling framework

The modelling framework, which was prototyped on the Namoi Basin in northern New South Wales, Australia, consists of two major components:

- 1) An upland catchment model, and
- 2) An instream sediment transport model.

The upland model simulates stream flow, suspended sediment and associated phosphorus using rainfall data, and is calibrated to daily stream flow time-series under historical conditions. The instream model routes suspended sediment from the outlet of upland catchments to gauging points downstream. The instream transport model can infer sources (re-suspension and bank erosion) and sinks (deposition) within a reach. Aerial photographs are used to assess the on-site effects of climate and land cover/use on erosion and the drainage network. Changes in land cover/use and the effects on the drainage network are related to the parameters in the rainfall-runoff model so that associated effects on stream flow and hence suspended sediment can be assessed.

The advantages of this approach are that it is temporally continuous and the time-step can be selected short enough for the model to be event responsive; it deals with gully and stream bank erosion as the predominant sources of sediment, that can be combined with interrill and rill erosion models; and it is based on an intermediate scale (of the order of 100 km²) for validation, but allows subscale variations in land cover/use to affect parameters at the intermediate scale, and it uses routing to obtain stream flow, suspended sediment and associated phosphorus concentrations at larger scales (avoiding the scaling up problem of a plot-scale approach). The whole methodology is data driven, the perceived weakness being that it cannot be applied immediately to other cases (Jakeman, Green, Beavis, Zhang, Dietrich and Crapper, 1999).

Although the framework is largely conceptual, the model structures approach can serve as an example of linking model components for catchments with gully and stream bank erosion as the predominant sources of sediment, and the use of aerial photographs to refine the sediment predictions with the hydrological model component.

7. CONCLUSIONS

7.1 IDENTIFICATION OF KNOWLEDGE GAPS AND RESEARCH NEEDS

The following knowledge gaps in the modelling ability were identified:

- A limitation in almost all the sediment yield models is that they do not consider shallow subsurface redistribution of water. Therefore, they are unable to identify hydrological active areas due to saturation excess flow or exfiltration of throughflow (interflow). Sediment production, and delivery of sediments and associated pollutants to watercourses, can be considerable in catchments where saturation excess overland flow and interflow are the dominant runoff generation mechanisms and these hydrological active areas are in close proximity to the watercourses. Furthermore, topographic convergence and subsurface hydrology are related to the amount of sediment and pollutant delivery to watercourses due to their contribution to the initiation and development of gullies. The inclusion of interflow and saturation excess flow runoff generation routines in sediment yield models will provide the basic structure to predict the areas of potential gully initiation and will give the models greater versatility as predictive tools when linking with suitable routines that model gully initiation and development.
- Both conceptual and physically-based sediment yield models address mainly interrill (sheet) and rill erosion, whilst ephemeral and classical gully erosion are rarely included as a model routine. Gully erosion can represent an important sediment source in a range of environments, and in many cases damage (sediment and pollutants) to watercourses by runoff from agricultural land relates to active gully erosion. Furthermore, gullies are effective links for transferring runoff, sediment and non-point source pollutants from hillslopes to watercourses, hence increasing the connectivity between the hillslopes and watercourses. There is a particular need for modelling studies of gully erosion to predict impacts of agriculture on gully erosion rates at various temporal and spatial scales, and their impacts on the hydrological and sediment yield processes at the hillslope- and catchment-scale.
- Several erosion studies have shown that the spatial and temporal distribution as well as the quality of deposited sediment in catchments under various agricultural management practice scenarios could not be predicted with a high degree of accuracy due mainly to the abovementioned knowledge gaps. This is especially true for gullied catchments since it is through gullies that a large fraction of sediment and non-point source pollutants within a catchment is redistributed and delivered to watercourses. The spatial and temporal distribution of sediment production and deposition are key processes for understanding sediment delivery from agricultural land to streams, and the transport of sediment from streams to large rivers and impoundments.
- Most catchment scale conceptual models (10-1000 km²) are forced to generalise the farming system as a form of land use or vegetation type and lack the capacity to simulate the impact of changed agronomic practice such as different rotations, lengths of fallow, tillage practices, stocking rates and grazing management on the sediment delivery from agricultural land to watercourses in detail. Therefore, current conceptual catchment models are ideal to identify problem areas and problematic land uses in a catchment requiring more rigorous studies, but should be supported with more detailed field scale models in selecting best management practices and erosion control measures.
- The value of field scale (<10 km²) physically-based models lies in the fact that the severity of sediment production and delivery under different agricultural management practices could be

studied in great detail in order to select best management practices and erosion control measures. However, they are generally too demanding in data requirements, and often in computing time, to be widely applicable. Lack of soil, vegetation and land use information at the detail scale is also severely limiting our ability to predict the impact of agricultural management practices in detail, and lack of knowledge of spatial variation of soils properties is proving critical to the effective application of detailed sediment yield models.

7.2 INTEGRATED MODELLING FRAMEWORK

The following issues related to the state of the art of modelling sediments in agriculture from field to catchment scale were identified from this literature study:

- The integration of a catchment scale and field scale modelling framework allows changes in the dominant processes in predicting the impact of agricultural practices on the sediment production and delivery in a catchment to be taken into account, while keeping within reasonable computational limits and data requirements. The catchment scale model generally will integrate over the details of micro-topography and the variability of soil and vegetation properties, and simplifies the detailed structure of the hydrograph to make use of available data at the broader scale. It is used mostly to identify problem areas and problematic land uses in a catchment requiring more rigorous studies. The field scale model considers more detailed processes in order to identify hydrologically active areas for sediment production and delivery and potential gully initiation to study the severity of sediment production and delivery under different agricultural management practices in greater detail in order to select best management practices and erosion control measures.
- Most of the integrated modelling frameworks are embedded in a GIS system which analyses digital elevation models (DEMs), remotely sensed vegetation/land cover maps and thematic soil and geology maps in order to subdivide a large catchment into sub-catchments of 5-10 km², and the sub-catchments be broken down into a series of representative hillslopes.
- Integrated modelling frameworks adopt a core catchment (terrestrial) hydrological model which is most often linked with an instream sediment routing model.
- The field scale model is most often nested within the catchment scale model through a catchment routing framework. The field scale model should preferably be embedded (models linked in parallel for model feedbacks) within the catchment scale model.
- It would be advantageous if the core catchment model had the ability to incorporate relevant and appropriate issue-specific and/or research models. The incorporation of other issue-specific and/or research models, such as gully erosion models/mapping, can be done by taking the output from the specific model as input for the core model, most often at the instream sediment routing model. The ability of the instream sediment routing model to accommodate for sediment supply from gullies, as predicted by gully erosion models/mapping, in addition to that supplied from the catchment model, will be an important model feature in catchments with gullies as the predominant sources of sediment.
- The integrated modelling framework should follow a minimum information requirement approach where the simplest model structure that satisfies the modelling needs of the policy maker while ensuring that the dominant erosion processes and factors are accounted for in that specific catchment. The information requirement will increase with the needs of the policy maker from the identification of problem areas and problematic land uses, prioritising of sources and management strategies and selection of best management practices; from rainfall excess, saturated excess and interflow dominated overland flow conditions; and from ungullied to (active) gullied catchments.

8. RECOMMENDATIONS

8.1 MONITORING

Establishing a limited set of representative catchments monitored for the purposes of non-point source processes conceptualisation, and non-point source model verification and development for a period of, say, five years, including, as far as possible, existing DWAF monitoring points. Monitoring of representative catchments on a regular basis, with reliable data acquisition and storage, should represent a nested system of a large catchment scale (10-1000 km²) - small catchment scale (0.5-10 km²) - hillslope and plot scale.

8.1.1 Sediment Yield Models

- Building simulation capability in current South African models, such as ACRU. The ACRU agrohydrological model can be improved by linking it with an instream hydraulic and sediment routing model, and by embedding a small catchment scale model which is able to identify hydrological active areas due to saturation excess flow and interflow.
- Deepening local experience in the application of comprehensive models with international standing, for example the HSPF, SWAT and MEDRUSH models. Local experience should also be gained in software packages such as BASINS which integrates a geographic information system, catchment land use and water quality monitoring data, and comprehensive water quality models into a single package.
- Evaluate models that consider interflow and saturation excess flow runoff generation routines and which provide the basic structure to incorporate gully erosion models/gully mapping, such as the Topog series of models.
- Evaluate gully erosion models such as the model developed by Sidorchuk for Australian conditions.
- Explore the possibilities that remote sensing hold for cost-effectively mapping gullies and gully growth.

9. REFERENCES

- Armstrong, J., Mackenzie, D. and Edwards, K. 1995. Catchment scale erosion prediction for land management planning. *Australian Journal of Soil and Water Conservation* 8(2), 29-40.
- Baldwin, D.S., Mitchell, A.M. and Olley, J.M. 2002. Pollutant-sediment interactions: sorption, reactivity and transport of phosphorus. In: P.M. Haygarth & S.C. Jarvis (eds.). *Agriculture, hydrology and water quality*. CABI Publishing, Wallingford, UK.
- Beckedahl, H.R. 1998. Subsurface soil erosion phenomena in South Africa. Justus Perthes Verlag Gotha, Germany.
- Bingner, R.L. 1990. Comparison of the components used in several sediment yield models. *Transactions of the American Society of Agricultural Engineers* 33(4), 1229-1238.
- Bocco, G. 1991. Gully erosion: processes and models. *Progress in Physical Geography* 15(4) pp. 392-406.
- Bocco, G. and Carcia-Oliva, F. 1992. Researching gully erosion in Mexico. *J of Soil and Water Conservation* pp. 365-369.
- Chadwick, D.R. and Chen, S. 2002. Manures. In: P.M. Haygarth & S.C. Jarvis (eds.). *Agriculture, hydrology and water quality*. CABI Publishing, Wallingford, UK.
- Chang, A.C. and Page, A.L. 2000. Trace elements slowly accumulating, depleting in soils. *California Agriculture* pp.49-55.
- Chartres, C.J. and Webb, A.A. 1998. Introduction: perspectives on the problem. In: J. Williams, A.H. Rosemary, L. Hester and L. Gascoigne (eds.). *Farming Action - Catchment Reaction: The effect of dryland farming on the natural environment*, pp. 308-320. CSIRO Publishing, Collingwood, Australia.
- Dallas, H.F. and Day, J.A. 1993. The effect of water quality variables on riverine ecosystems: A review. Water Research Commission report No. TT61/93.
- De Vries, W., Römkens, P.F.A.M, Van Leeuwen, T. and Bronswijk, J.J.B. 2002. Heavy metals. In: P.M. Haygarth and S.C. Jarvis (eds.). *Agriculture, hydrology and water quality*. CABI Publishing, Wallingford, UK.
- Department of Water Affairs and Forestry, South Africa. 1985. Management of the Water Resources of the Republic of South Africa. Department of Water Affairs, Pretoria.
- Department of Water Affairs and Forestry, South Africa. 1991. Water quality management policies and strategies in the RSA. Department of Water Affairs and Forestry.
- Endreny, T.A. and Wood, E.F. 1999. Distributed Watershed Modeling of Design Storms to Identify Non-point Source Loading Areas. *Journal of Environmental Quality* 28, 388-397.
- Elliot, B., Ward, A., Lane, L.J. and Laffen, J.M. 1989. An overview of WEPP: Water Erosion Prediction Project. Paper presented at the Southern African Regional Commission for Conservation and Utilisation of Soil, 16th regular meeting of the subcommittee for Land-use Planning and Erosion Control (LUPEC). Mbabane, Swaziland, 12-14 September 1989.

- Fitzhugh, T.W. and Mackay, D.S. 2001. Impact of subwatershed partitioning on modeled source- and transport-limited sediment yields in an agricultural nonpoint source pollution model. *Journal of Soil and Water Conservation* 56(2), 137-143.
- Foster, G. R. and Meyer, L.D. 1977. Soil Erosion and Sedimentation by Water – An Overview. In: *Proceedings of Soil and Sedimentation by Water*, pp. 1-13. American Society of Agricultural Engineers, Chicago, Illinois.
- Garland, G., Hoffman M.T. and Todd, S.W. 1999. Soil degradation. In: T. Hoffman, S. Todd, Z. Ntshona and S. Turner (eds.). *Land degradation in South Africa*. [Web] <http://www.nbi.ac.za/landdeg/>. [Date of access 20/12/2003].
- Geeson, N.A. and Thornes, J.B. 1996. The MEDALUS Project. Davis-Wise Design Company, Bristol, UK.
- Hanson, G.J., Robinson, K.M. and Cook, K.R. 2001. Prediction of headcut migration using a deterministic approach. *Transactions of the American Society of Agricultural Engineers* 44(3), 525-527.
- Hatton, T.J., Nicoll, C.L., Hairsine, P.B. and Cresswell, H.P. 1998. Models of catchment water quality and their ability to predict the consequences of changes in land use and management practices. In: J. Williams, A.H. Rosemary, L. Hester and L. Gascoigne (eds.). *Farming Action Catchment Reaction: The effect of dryland farming on the natural environment*, pp. 308-320. CSIRO Publishing, Collingwood, Australia.
- Haygarth, P.M. and Jarvis, S.C. 1999. Transfer of phosphorus from agricultural soils. *Advances in Agronomy* 66, 196-236.
- Herpe, Y.J.P., Troch, P.A., Quinn, P.F. and Anthony, S. 2002. Modelling hydrological mobilization of nutrient pollutants at the catchment scale. In: P.M. Haygarth and S.C. Jarvis (eds.). *Agriculture, hydrology and water quality*. CABI Publishing, Wallingford, UK.
- Herselman, J.E. and Steyn, C.E. 2001. Predicted concentration of trace elements in South African soils. Report on contract for the National Department of Agriculture by Institute for Soil, Climate and Water of the Agricultural Research Council. ISCW report No. GW/A/2001/14.
- Hook, R.A. 1998. Landscape structure and function - the fundamental causes of land and water degradation. In: J. Williams, A.H. Rosemary, L. Hester and L. Gascoigne (eds.). *Farming Action Catchment Reaction: The effect of dryland farming on the natural environment*, pp. 121-131. CSIRO Publishing, Collingwood, Australia.
- Hook, R.A., Fleming, M.R. and Thomas, C. 1998. Understanding Models. In: J. Williams, A.H. Rosemary, L. Hester and L. Gascoigne (eds.). *Farming Action - Catchment Reaction: The effect of dryland farming on the natural environment*, pp. 241-268. CSIRO Publishing, Collingwood, Australia.
- Jakeman, A.J., Green, T.R., Beavis, S.G., Zhang, L., Dietrich, C.R. and Crapper, P.F. 1999. Modelling upland and instream erosion, sediment and phosphorus transport in a large catchment. *Hydrological Processes* 13(5), 745-752.

Jewitt, G.P.W. and Görgens, A.H.M. 2000. Scale and model interfaces in the context of integrated water resources management for the rivers of the Kruger National Park: An integrated modelling system for predicting the impacts of changes in water quality and quality brought about by upstream development. Water Research Commission, Pretoria. WRC Report No. TT627/1/00.

Kienzle, S.W., Lorentz, S.A. and Schulze, R.E. 1997. Hydrology and Water Quality of the Mgeni Catchment. Water Research Commission Report. TT 87/97. Pretoria.

Kirkby, M.J., Imeson, A.C., Bergkamp, G. and Cammeraat, L.H. 1996. Scaling up processes and models from the field plot to the watershed and regional areas. *J. Soil and Water Conservation*. 51(5), 391-396.

Knisel, W.G. 1993. GLEAMS: Groundwater Loading Effects of Agricultural Management Systems. University of Georgia, Coastal Plains Experiment station, Biological and Agricultural Engineering Department Publication No. 5, 260 pp.

Lal, R. and Elliot, W. 1994. Erodibility and Erosivity. In: R. Lal (2nd edition). *Soil Erosion Research Methods*, pp. 181-208. St. Lucie Press, Florida.

Leinweber, P., Turner, B.L. and Meissner, R. 2002. Phosphorus. Heavy metals. In: P.M.Haygarth and S.C. Jarvis (eds.). *Agriculture, hydrology and water quality*. CABI Publishing, Wallingford, UK.

Lenzi, M.A., Agostino, V.D' and Billi, P. 1999. Bedload transport in the instrumented catchment of the Rio Cordon. Part 1: Analysis of bedload records, conditions and threshold of bedload entrainment. *Catena*. 36, 171-190.

Lorentz, S.A. and Schulze, R. E. 1995. Sediment yield. In: R. E. Schulze. *Hydrology and Agrohydrology: A Text to Accompany the ARCU 3.00 Agrohydrological Modelling System*. Water Research Commission. Pretoria, Report TT69/95. pp. AT16-1 to AT16-34.

McDowell, R.W., Sharpley, A.N., Kleinman, P.J.A. and Gburek, W.J. 2002. Hydrological source management of pollutants at soil profile scale. In: P.M.Haygarth and S.C. Jarvis (eds.). *Agriculture, hydrology and water quality*. CABI Publishing, Wallingford, UK.

McLaughlin, M.J., Kookana, R.S., Donnelly, T.H. and Wasson, R.J. 1998. Land degradation processes and water quality effects: organic matter, soil and nutrient loss, and chemical residues. In: J. Williams, A.H. Rosemary, L. Hester and L. Gascoigne (eds.). *Farming Action- Catchment Reaction: The effect of dryland farming on the natural environment*, pp. 191-214. CSIRO Publishing, Collingwood, Australia.

Merkel, W.H., Woodward, D.E. and Clarke, D. 1988. Ephemeral gully erosion model (EGEM). *Modelling Agricultural, Forest, and Rangeland Hydrology*. American Society of Agricultural Engineers Publication 07-88, 315-323.

Meyer, L.D. and Wischmeier, W.H. 1969. Mathematical simulation of the process of soil erosion by water. *Transactions of the American Society of Agricultural Engineers* 12: 754-762.

Midgley, D.C. 1952. A preliminary survey of the surface water resources of the Union of South Africa. Unpublished PhD thesis. University of Natal, Pietermaritzburg.

Morgan, R.P.C. 1995. *Soil erosion and Conservation* (2nd edition). Longman Group Limited, Essex, England.

Mutchler, C.K., Murphree, C.E. & McGregor, K.C., 1994. Laboratory and field plots for erosion research. In: R. Lal (2nd edition). *Soil Erosion Research Methods*, pp. 181-208. St. Lucie Press, Florida.

Nash, D., Halliwell, D. & Cox, J., 2002. Hydrological mobilisation of nutrient pollutants at the field/slope scale. In: P.M. Haygarth & S.C. Jarvis (eds.), *Agriculture, hydrology and water quality*. CABI Publishing, Wallingford, UK.

Nearing, M.A., Lane, L.J. and Lopes, V.L., 1994. *Modelling Soil Erosion*. In: R. Lal (2nd edition). *Soil Erosion Research Methods*, pp. 181-208. St. Lucie Press, Florida.

Onstad, C. A., Mutchler, C.K. and Bowie, A.J., 1977. Predicting sediment yields. In: *Proceedings of Soil and Sedimentation by Water*, pp. 43-58. American Society of Agricultural Engineers, Chicago, Illinois.

Pegram, G.C., Gorgens, A.H.M. and Ottermann, A.B., 1997. A framework for addressing the information needs of catchment water quality management. *Water SA* 23(1), 13-20.

Pegram, G.C. and Gorgens, A.H.M., 2001. A guide to non-point source assessment: To support water quality management of surface water resources in South Africa. Water Research Commission, Pretoria. WRC Report No. TT142/01.

Poesen, J.W., Boardman, J., Wilcox, B. and Valentin, C., 1996. Water erosion monitoring and experimentation for global change studies. *J. of Soil and Water Cons.* 51(5), 386-390.

Poesen, J., Nachtergaele, J., Verstraeten, G. and Valentin, C., 2003. Gully erosion and environmental change: importance and research needs. *Catena*, 50, 91-133.

Quinton, J.N., 1997. Reducing predictive uncertainty in model simulations: a comparison of two methods using the European Soil Erosion Model (EUROSEM). *Catena*, 30, 101-117.

Rooseboom, A., 1978. Sedimentafvoer in Suider-Afrikaanse riviere/Sediment discharge in Southern African rivers. *Water SA* 4, 14-17.

Rooseboom, A., 1992. Research Project – Preparation of an overview document on sediment transport in South Africa, including revision of the sediment yield map of Southern Africa. Water Research Commission, Pretoria. WRC Report No. TT297/1/92.

Rooseboom, A., Verster, E., Zietsman, H.L. and Lotriet, H.H., 1992. The development of the new sediment yield map of Southern Africa. Water Research Commission, Pretoria. WRC Report No. TT297/1/92.

Rowntree, K.M. and Wadeson, R.A., 1999. A hierarchical geomorphological model for the classification of selected South African rivers. Water Research Commission, Pretoria. WRC Report No. 497/1/99.

Schmidt, E.J., Smithers, J.C., Schulze, R.E. and Mathews, P., 1998. Impacts of sugarcane production and changing land use on catchment hydrology. Water Research Commission, Pretoria. WRC Report No. 419/1/98.

- Schoeman, J.L. and Scotney, D.M. 1987. Agricultural Potential As Determined by Soil, Terrain and Climate. *South African Journal of Science* 83:260-268.
- Schulze, R.E. 1995. Streamflow. In: R. E. Schulze. Hydrology and Agrohydrology: A Text to Accompany the ARCU 3.00 Agrohydrological Modelling System. Water Research Commission. Pretoria, Report TT69/95. pp. AT16-1 to AT16-34.
- Schulze, R.E., Angus, G.R., Lynch, S.D. and Smithers, J.C. 1995. ACRU: Concepts and structure. In: R. E. Schulze. Hydrology and Agrohydrology: A Text to Accompany the ARCU 3.00 Agrohydrological Modelling System. Water Research Commission. Pretoria, Report TT69/95. pp. AT16-1 to AT16-34.
- Schumm, S.A. 1977. Fluvial Systems, John Wiley, New York.
- Schwartz, H.I. and Pullen, R.A. 1966. A guide to the estimation of sediment yield in South Africa. Civil Engineering in South Africa, December, 343-346.
- Selby, M.J. 1993. Hillslope materials and processes (2nd edition). Oxford University Press, New York.
- Sidorchuk, A. 1998. A dynamic model of gully erosion. In: Boardman, J., Favis-Mortlock, D. (Eds.), Modelling Soil Erosion by Water. NATO-Series 1, vol. 55, 451-460. Berlin, Heidelberg.
- Sidorchuk, A. and De Roo, A.P.J. 1999. Dynamic and static models of gully erosion. *Catena* 37: 3-4, 401-414.
- Sidorchuk, A., Marker, M., Moretti, S. and Rodolfi, G. 2003. Gully erosion modelling and landscape response in the Mbuluzi River catchment of Swaziland. *Catena* 50, 507-525.
- Smith, R.E., Goodrich, D.C. and Quinton, J.N. 1995. Dynamic, distributed simulation of watershed erosion: The KINEROS2 and EUROSEM models. *Journal of Soil and Water Conservation* 50(5), 517-520.
- Snaddon, C.S. 1999. Degradation of surface water resources. In: T. Hoffman, S Todd, Z Ntshona and S. Turner (eds.). Land degradation in South Africa. [Web] <http://www.nbi.ac.za/landdeg/>. [Date of access 20/12/2003].
- Van Rensburg, J.D.J. and Dent, M.C. 1997. Development of a Water Quality and Quality Modelling System which will provide a common currency for communication between researchers in the Kruger National Park Rivers Research Programme. Water Research Commission, Pretoria. WRC Report No. 654/1/97.
- Walling, D. E. 1994. Measuring sediment yield from river basins. In: R. Lal (2nd edition). Soil Erosion Research Methods, pp. 39-80. St. Lucie Press, Florida.
- Walter, M.T., Walter, M.F., Brooks, E.S., Steenhuis, T.S., Boll, J. and Weiler, K. 2000. Hydrologically Sensitive Areas: Variable Source Area Hydrology Implications for Water Quality Risk Assessment. *J. Soil and Water Conservation*. 55(3), 277-278.
- Wang, E., Buland, D., Dayton, B. and Diluzio, M. 2000. SSURGO County-Level Soils Data Usage with the SWAT Watershed Model. *J. of Soil and Water Cons.* 55(3), pp. 417.

Wasson, R.J. 1998. Dryland farming, erosion and stream sediments – the problem of catchment scale. In: J. Williams, A.H. Rosemary, L. Hester and L. Gascoigne (eds.). *Farming Action - Catchment Reaction: The effect of dryland farming on the natural environment*, pp. 215-228. CSIRO Publishing, Collingwood, Australia.

Wijdenes, D.J.O., Poesen, J., Vandekerckhove, L. and Ghesquiere, M. 2000. Spatial distribution of gully head activity and sediment supply along an ephemeral channel in Mediterranean environment. *Catena* 39, 147-167.

Williams, J.R. and Berndt, H.D. 1977. Sediment yield based on watershed hydrology. *Transactions of the American Society of Agricultural Engineers* 20, 1100-1104.

Woodward, D. E. 1999. Method to predict cropland ephemeral gully erosion. *Catena* 37, 393-399.

DISCUSSION PAPER 4:
WATERBORNE PATHOGENS IN AGRICULTURE

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WATER BORNE PATHOGENS IN AGRICULTURE

1. INTRODUCTION

1.1 INTRODUCTION TO THE POSITION PAPER

It is increasingly recognised that non-point source (NPS), or diffuse, pollution plays a major role in the degradation of water quality; specifically with respect to salinity, eutrophication (nutrient enrichment), sediments, pathogens, pesticides and some heavy metals. Since most of the land area in South Africa is utilised for agricultural activities, agriculture has both locally and internationally been implicated as a major source of NPS pollution. It is therefore necessary to assess the contribution that the different agricultural activities make to the different symptoms of NPS pollution, to devise the approaches through which these can be controlled and to determine and predict the effect that control measures will have on reducing NPS pollution. In 2003 the Water Research Commission initiated a one-year study to develop a research programme that will address those issues that require priority attention. A workshop on NPS impacts of Agriculture that was held in June 2002 and the priority issues were identified as pesticides, nutrients, sediment and metals, salinity and pathogens. The first step in this study was to take stock of the present situation and existing knowledge base with regard to these five NPS concerns.

This paper is designed to provide a reconnaissance level summary of the knowledge base about waterborne pathogens in agriculture. It was decided to investigate waterborne pathogens in less detail than other NPS concerns because agriculture was not regarded as a major contributor when compared to other nonpoint sources of waterborne pathogens such as dense settlements and urban runoff. During the design of the National Microbiological Monitoring Programme, potential high health risk areas were identified (Van Niekerk, 2000). The potential health risk was a function of two factors, land use causing faecal pollution, and water use sensitive to faecal pollution. The technical team assigned weights to different landuses to indicate their perception of the importance of different sources. Intensive farming enterprises were assigned a weight of 3% compared to human settlements with no sanitation infrastructure (53%) and human settlements with inefficient sanitation infrastructure (43%). This led to the conclusion that pathogens from agricultural sources were not regarded at a national scale, as the major source of pathogen pollution.

The team was requested to address the following aspects in each position paper:

- A brief overview of the extent of the problem within a national context and statements on data and information constraints.
- A description of the origins and sources of the problem: identify the main sources of the specific agricultural NPS problem.
- A description of the transfer/transport/transformation processes and mechanisms.
- A description of models that simulate the production, transport and fate of the pollutants to assess the predictive (modelling) ability that is available to address each of the specific agricultural NPS problems and identification of knowledge gaps.
- Mitigation measures/best management practises to address the problem: assess the state of mitigation measures to address the specific agricultural NPS problem and identification of knowledge gaps.

1.2 INTRODUCTION TO WATERBORNE PATHOGENS IN AGRICULTURE

Pathogens from agriculture can serve as a source of contamination of water. A pathogen is an agent that can cause diseases in animals and plants. Waterborne zoonotic disease is a term that is used to describe illness that is transmitted among animals and humans by water (Rosen, 2000). The transmission from livestock to humans is of particular concern in this paper. Most waterborne pathogens occur in human and animal faeces and enter water either by defecation in streams and other waterbodies, or carried by overland flow and/or subsurface flow. In the USA, agriculture is regarded as a major cause of not meeting water quality standards for indicator organisms (Rosen, 2000). In South Africa the situation is unclear because the focus in microbial monitoring system design has been on monitoring the effects of poor sanitation and human settlements rather than the identification of all potential sources. This is a pragmatic solution due to the difficulties in sampling and analysing microbial samples and the focus at a national level on water supply and sanitation issues.

The principle pathogens of concern to agriculture are those that can cause the outbreak of waterborne diseases. These pathogens have the following characteristics:

- Organisms are released into the environment in large numbers, or are highly infectious to humans and animals at low doses.
- They can survive and remain infectious for long periods or they are resistant to water treatment, and
- Some types of bacterial pathogens can multiply outside of a host under favourable environmental conditions.

From an agricultural nonpoint source perspective, the key waterborne pathogens of concern have been identified as:

- Protozoa (*Giardia* and *Cryptosporidium*)
- Bacteria that can infect multiple species (*Escherichia coli*, *Campylobacter* & *Salmonella*)
- Bacterial infections associated with humans (*Shigella*)
- Human viruses (e.g. Hepatitis)
- Cyanobacteria and other toxic algal blooms
- Helminths

2. BRIEF OVERVIEW OF THE EXTENT OF THE PROBLEM

2.1 PROTOZOA

The protozoa of primary concern are *Cryptosporidium* and *Giardia*. The protozoa cause mild to severe diarrhoea and can potentially affect the lifespan of immune-compromised individuals such as those suffering from AIDS. In South Africa 11 out of 110 children who were infected with cryptosporidiosis died and relatively high mortalities in patients with diarrhoea due to *Cryptosporidium* have been reported (in Jarmey-Swan *et al.*, 2001).

These protozoa are found throughout the world and are commonly transmitted from faeces to mouth in animals and humans. They cannot reproduce outside the host organism. In natural undisturbed catchments, water can be contaminated with protozoa that originate from wildlife in the area.

Cryptosporidium is a protozoan parasite that infects humans, livestock (cattle, sheep, goats, pigs & horses), domestic animals (cats and dogs) and wildlife. *Cryptosporidium* oocysts are released in the faeces of an infected host and are immediately infective. The thick walled oocysts are environmentally resistant and can survive for as long as 180 days in water (Rosen, 2000).

Giardia is a protozoa that infects the intestinal tracts of mammals such as humans, dogs, cats and wildlife. *Giardia* cysts are more widespread than *Cryptosporidium* (Rosen, 2000), and observations in Namibia seem to confirm this for Southern Africa (Menge *et al.*, 2001). *Giardia* cysts are released in the faeces of an infected host, it is not immediately infective after release but it is resistant and can survive for long periods in the environment, especially at low water temperatures. There is considerable controversy whether *Giardia* cysts from livestock can infect humans (Rosen, 2000). Newman *et al.* (2003) argue that there is no scientific evidence that cattle-derived oocysts/cysts in surface water can survive and infect humans at pathogenic levels even though there appears to be a link between cattle and the incidence of cryptosporidiosis cases in Canada (where their study was undertaken).

In the USA, most of the outbreaks of waterborne cryptosporidiosis have been ascribed to sewage contamination and inadequate drinking water treatment. Contamination of water by farm animals was only reported in three cases (Rosen, 2000).

In Southern Africa, Jarmey-Swan *et al.* (2001) failed to find a correlation between the occurrence of *Cryptosporidium* and *Giardia* related diarrhoea in KwaZulu-Natal and climatic factors. The assumption was that runoff from faecally contaminated land or farm slurry can contaminate water resources resulting in infection of those using the water further downstream. They found that although the prevalence of diarrhoea and infection increased when rainfall increased, it was not a statistically significant correlation with climatic factors. They concluded that other factors such as personal hygiene and sanitation also contributed to infection. In Namibia, Menge *et al.* (2001) monitored the occurrence of *Giardia* and *Cryptosporidium* in Goreangab Dam outside Windhoek. The dam receives mostly treated wastewater effluent and infrequent urban runoff. They found that protozoa occurred in about 60% of the samples from the dam and about 55% of the samples from the maturation pond effluent. Du Preez and Gericke (1998) monitored drinking water sources of semi-urban and rural communities north of Pretoria and found that about 17% of the samples were contaminated with *Giardia* cysts and 8% with *Cryptosporidium* cysts. *Giardia* cysts showed no seasonal differences but higher counts of *Cryptosporidium* cysts were found during the summer rainfall months.

2.2 BACTERIA

Bacteria are widespread and mostly beneficial for decomposing dead plant material and releasing nutrients back into the environment. This paper will therefore focus on a few bacteria that know to be harmful to humans and livestock. It is difficult to identify and count individual bacteria and in the water field has been common practice to culture key organisms that are associated with faecal contamination. These organisms are regarded as indicators of the possible presence of pathogens in the water and a risk of disease.

Escherichia coli - The most commonly used indicator bacterium that is also associated with gastrointestinal illness is *Escherichia coli* or *E.coli*. It is a member of the faecal coliforms group and has been shown epidemiological to cause gastrointestinal symptoms. *E.coli* O157:H7 is a potentially deadly bacteria that can cause bloody diarrhoea and dehydration. Young children, the elderly and immuno-compromised individuals are most susceptible. Infection is usually through eating food or drinking fluids containing the bacteria. The organism is not pathogenic to cattle but livestock is often cited as a source of *E.coli*, especially feedlots.

Campylobacter - Several species of *Campylobacter* can cause infection in humans but it occurs as sporadic cases rather than outbreaks involving large numbers of people. The primary route of infection appears to be the ingestion of contaminated foods (poultry or raw milk), intake of untreated surface water and contact with pets suffering from diarrhoea. Cattle and poultry faeces have been shown to contain *Campylobacter* sp. but Rosen (2000) state that there were very few cases of human infections that could be traced back to livestock manure or effluent.

Salmonella - *Salmonella* species cause diarrhoea and systemic infections that can be fatal to susceptible individuals. The majority of cases in the USA are associated with foodborne illness. Foods implicated include poultry and poultry products, dairy products, eggs, seafood and fresh produce. There is uncertainty whether manure or poultry effluent is a significant source of waterborne salmonellosis in humans (Rosen, 2000).

Other waterborne bacteria associated with agricultural activities are of lesser concern. These include *Yersinia*, *Brucella* sp. and *Leptospira interrogans*.

2.3 VIRUSES

In the USA there appears to be little evidence that viruses shed in the excrement of livestock posed a waterborne threat to human health. It appears that runoff from treated sewage sludge or seepage from septic tanks may be the most significant source of pathogenic viruses. Livestock derived viruses are generally not considered a source of human waterborne infection (Rosen, 2000).

2.4 TOXIC ALGAL BLOOMS

Algae occur naturally in the water environment but under certain conditions, toxic algal blooms can develop as a result of nutrient enrichment associated with agricultural activities. Although the position paper on nutrients deals with the effects of nutrient enrichment as a result of agricultural activities, this section will only describe the occurrence and effect of toxic algal blooms. Harding and Paxton (2001) provide an excellent review of cyanobacteria or blue-green algae in South Africa. A characteristic of cyanobacteria is the ability to produce a range of extremely potent biological toxins. The algal genera associated with toxins that most commonly occur in South African water bodies are *Anabaena*, *Microcystis*, *Oscillatoria* (*Planktothrix*), *Aphanizomenon* and *Nodularia*. In general, toxins are produced in a random and unpredictable fashion (Harding & Paxton, 2001). There are examples

where toxic algal blooms occurred in reservoirs where the dominant source of nutrients was agriculture, and resulting in the death of livestock. These include Allemanskraal Dam and Erfenis Dam in the Vet River system where *Microcystis* and *Anabaena* were identified as the causative organism (Harding & Paxton, 2001, Van Ginkel *et al.*, 2000). The most severe case of livestock loss associated with toxic algae occurred in 1996 in the Kareedouw district where 290 in-milk dairy cows died and a further 70 animals had to be slaughtered after drinking from a small cement lined reservoir that showed the presence of toxic algae.

However, in most reservoirs where cyanobacteria blooms occur, the source of nutrient enrichment could be traced to loadings from wastewater treatment works in the catchment.

2.5 HELMINTHS

Helminths are worms that may be free-living or parasitic in plants and animals. Parasitic worms like flukes, tapeworms and roundworms are of greatest concern. *Ascaris lumbricoides* is the biggest roundworm and the most common parasite found in humans. Adult worms live in the small intestines and eggs are passed in the faeces. The eggs can contaminate crops when irrigation water has not been adequately disinfected. Sewage sludge can also be a source of contamination if the sludge has not been treated adequately. Humans are infected if they eat raw produce that is contaminated with live *Ascaris* eggs. There appears to be a risk for organically produced crops if special care is not taken with the manure that is used (Nelson, 1997).

3. DESCRIPTION OF ORIGINS AND SOURCES

3.1 SOURCES OF WATERBORNE PATHOGENS

There are several pathways for pathogens to enter surface water sources (Rosen, 2000) and agriculture is just one the sources (Figure 3.1).

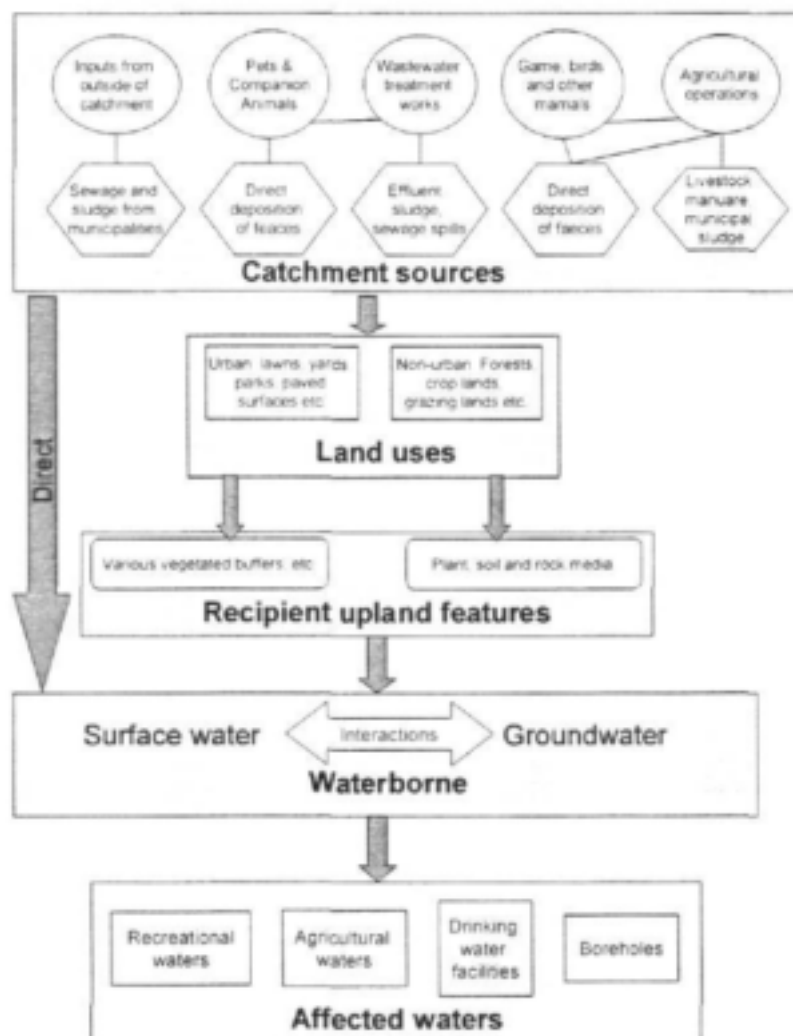


Figure 3.1 Potential pathways for waterborne pathogen movement in catchments

In agriculture, water plays a major role in mobilising and transporting pathogens. Rainfall washes pathogens from faeces or vegetation into the soil or as surface runoff into surface water streams and rivers.

Intensive livestock farming enterprises (feedlots, piggeries etc.) pose the greatest risk of pathogen contamination. In South Africa, DWAF uses the guide developed by the WRC (Funke *et al.*, 1984) for the handling of manure from intensive animal feeding units to control pollution from feedlots, piggeries etc. For free-ranging livestock, bare areas such as kraals also pose a high potential for pathogen runoff into surface water or leaching into groundwater. Cattle drinking directly from surface water streams and dams and defecating in the water is an obvious source of contamination. For example, in

May and June 2000 there was an outbreak of gastroenteritis in Walkerton, Canada. During this outbreak over 2300 cases were reported and six people died as a result of the outbreak. The outbreak was traced to *E. coli* O157:H7 contamination of wells, used for drinking water supply, and situated close to 13 livestock farms. Pathogen originating from manure entered the wells during heavy rains (Bruce-Grey-Owen Sound Health Unit, 2000), resulting in the outbreak.

Land application of sewage and sludge can act as concentrated sources of pathogens and requires proper treatment, handling and application methods to minimize the infection of humans and animals. Some pathogens can survive the sewage treatment process even though their numbers are reduced. In South Africa the use of sewage sludge for agricultural purposes is guided by a set of guidelines developed by the Department of Health (1978) and the addendum to the guidelines.

Domestic animals are also a source of pathogen contamination in urban areas. They are a potential reservoir of a number of pathogens including *Giardia*, *Cryptosporidium* and *Salmonella*. Domestic animal waste in yards and walkways next to streams are subject to direct runoff and can contaminate surface water streams in urban areas.

Wildlife also acts as pathogen reservoirs and their wastes enter surface water streams during rainfall runoff events. Waterfowl can also add to the pathogen loads in dams and rivers where they nest and feed.

3.2 SURVIVAL OF PATHOGENS IN THE ENVIRONMENT

Certain pathogens exit in the host animal in forms that can survive for an extended period of time in the environment (Table 3.1). This allows the pathogen an opportunity to be ingested again by another host and to repeat its life cycle. Survival of pathogens outside the host is dependent on a number of factors that include temperature, soil types, moisture content, sunlight, etc.

Table 3.1 Survival times (days) of pathogens in various media (Rosen, 2000)

Pathogen	Slurry	Faecal paddies	Soil	Water	Plants
Bacteria					
<i>Salmonella</i> sp.	250+	200+	150+	16	
<i>E. coli</i>	300+	200+	200+	35	
General maxima*			1 year		180
Viruses					
General maxima			1 year		60
Protozoan cysts					
General maxima				180+	5 (dry)
Helminth ova					
General maxima			7 years		150 (dry)

* General maxima is the longest time that the pathogen can survive.

4. DESCRIPTION OF MODELS

Most models of bacterial die-off are based on Chick's Law, a simulation of a simple first-order reaction in chemical kinetics given as (Rosen, 2000, Salter *et al.*, 2003a):

$$\frac{N_t}{N_0} = 10^{-kt}$$

where N_t = number of bacteria at time t , N_0 = number of bacteria at time 0, t = time in days, and k = first order or die-off rate constant.

Salter *et al.*, (2003) reviewed the application of Chick's Law to modelling microbial water quality as well as a number of variations on the model. They also reviewed different methods for estimating the die-off constant, k , as a function of time, as a function of temperature, pH and oxidation potential and as a function of pH, dissolved oxygen, nutrient concentrations and temperature and functions that include solar and UV radiation.

Shoemaker *et al.* (1997) undertook a comprehensive review of models that can be used for catchment assessment studies and they classified models into four main categories and in some cases, subcategories:

- Watershed-scale loading models (simple methods, mid-range models, detailed models)
- Field-scale loading models
- Receiving water models (Hydrodynamic models, steady-state water quality models, dynamic water quality models, mixing zone models)
- Integrated modelling systems

The same classification system will be used in this report except the term 'catchment' rather than 'watershed' will be used to stay consistent with terminology in South Africa. This section will only refer to models that are used to simulate the loading, transport and fate of pathogens. It is assumed that models that simulate toxic algae are dealt with in the nutrients position paper.

4.1 CATCHMENT-SCALE LOADING MODELS

Catchment-scale loading models refer to models that can be used to assess the effects of land use and farming practices on pollutant loadings to waterbodies. They can be classified as:

- Simple – used for rapidly identifying critical areas with minimal effort and data. These typically rely on empirical relationships between physiographic characteristics of the catchment and pollutant export.
- Mid-range models – used to assess pollutant sources and impact on a broad geographic scale. They are typically designed to interface with GIS that facilitates the selection of parameter values.
- Detailed models – These models best represent the current understanding of catchment processes that affect pollution generation. They provide a detailed and accurate prediction of flows and water quality at different points in a catchment. They are, however, often costly in terms of time and resources to apply.

Examples of catchment-scale loading models include the following:

IMPAQ – IMPAQ is a catchment scale monthly water quality model that uses the flows generated by the water resources planning model (WRPM) and the water resources yield model (WRYM) to simulate the export of constituents from a catchment, the transport in rivers and streams and the fate in reservoirs. The model simulates *E.coli* counts in addition to salts, phosphorus and suspended sediment (DWAF, 1998). The model was applied to the Amatole system to evaluate the water quality impacts of different system operating scenarios.

WEND & WEBD – Cassell *et al.* (2002) describes a Watershed Ecosystem Nutrient Dynamics (WEND) model and Watershed Ecosystem Bacterial Dynamics (WEBD) model that was developed with mass balance concepts to estimate contaminant export from catchments over long and short-term futures. An objective of the models is to evaluate policies for catchment management. The models were developed with the STELLA interactive object-oriented software package and were tested on three catchments; a dairy/urban catchment, a poultry/urban catchment and a swine dominated catchment.

Process based modelling

Salter *et al.* (2003b) reviewed a process based model developed by Jenkins *et al.* (1984) which is based on the concept of a catchment being a "multiple storage and release system" for bacteria. The catchment land surface, river water and river sediments are treated as bacteria stores. The model attempts to integrate transport mechanisms in the catchment and the river with die-off in the river. Salter *et al.* (2003b) provides a good review of some of the individual equations and approaches used in the process based model.

4.2 FIELD-SCALE LOADING MODELS

Field-scale loading models refer to models that represent smaller, homogenous areas in more depth. They are useful for assessing management practices on a microscale. Field scale models are often used as part of a nested modelling approach where the results of field scale models are used in larger catchment scale models.

Shoemaker *et al.* (1997) lists three examples of field scale models commonly used in the USA for estimating sediment and other constituent loads. These are CREAMS/GLEAMS, Opus, and WEPP (Water erosion prediction project). None of these models simulate bacterial or pathogen loads at a field scale. No other field scale models that simulate pathogen export could readily be identified. Rosen (2000) referred to farm scale model (MWASTE/MSPRED) that was developed by the National Water & Climate Centre of the USDA to estimate indicator bacterial loads at a farm scale as a result of manure management practices. No additional information or reference could be obtained on the model.

4.3 RECEIVING WATER MODELS

Receiving water models refer to models that are used to examine the interactions between loadings and the response of the river or reservoir system. They vary widely in complexity depending on the processes being simulated. These include hydrodynamic and water quality models. Examples of receiving water models include:

QUAL2E stream water quality model - The QUAL2E model is a steady state stream water quality model that can simulate 15 water quality constituents, one of which is coliforms (Brown & Barnwell, 1987). In South Africa, Venter *et al.* (1996) evaluated the use of the QUAL2E river water quality model to simulate microbial water quality in the Rietspruit. They found that the model was useful as a management tool in highly contaminated catchments.

QUASAR - PC-QUASAR is a river network water quality and flow model for river networks developed by CEH Wallingford, formerly the Institute of Hydrology (<http://www.nwl.ac.uk/ih/www/aboutih/maboutih.html>). It is designed to be used by catchment management authorities and water/sewerage utility companies to help manage river water quality. PC-QUASAR describes the changes in water quality over time and allows tracking of pollution pulses downstream. Parameters modelled are flow, nitrate, dissolved oxygen, ammonia, temperature, *E. coli*, pH, biochemical oxygen demand and a conservative pollutant or tracer. Salter *et al.* (2003b) reviewed the *E. coli* model in QUASAR. Die-off is modelled with a 1st order temperature dependant decay rate and the model only describes decay downstream of a point source of bacteria (like a feedlot in the case of an agricultural source).

CAEDYM - Hipsey (2002) described how a three-dimensional reservoir water quality model was modified and used to simulate the fate and transport of *Cryptosporidium* oocysts in the Myponga Reservoir, a drinking water reservoir in Australia. The modelling team included a *Cryptosporidium* fate module in the CAEDYM aquatic ecology and water quality model to consider the effects of temperature and UV light on oocyst inactivation in addition to settling and resuspension of oocysts. The model is used in a descriptive mode to describe the distribution of *Cryptosporidium* once they enter a reservoir and it is used in an operational mode to predict when potential pathogen problems may occur at the water offtake point under high inflow conditions.

Neural network modelling – Neelakantan *et al.* (2001) developed a neural network modelling system for *Cryptosporidium* and *Giardia* in the Delaware River, USA. They concluded that neural networks could learn to identify a unique set of water quality characteristics associated with peak concentrations of *Cryptosporidium* and *Giardia*. They found that *Clostridium perfringens* and *E. coli* were two crucial input parameters for predicting the protozoa concentrations.

Others receiving water models that include state variables for pathogens include CE-QUAL-W2 and CE-QUAL-RIV1 of the US Army Corps of Engineers, HSPF of the USEPA, and RWQM1 of the IWA Task Group on River Water Quality Modelling.

4.4 INTEGRATED MODELLING SYSTEMS

Integrated modelling systems often represent the newer generation modelling systems that are easier to use by adding Windows based pre- and post-processors, linked watershed and response models, linked to databases and modular in design. Examples of the new generation of integrated modelling systems that have a waterborne pathogen component, include:

PC-VirGIS – Virginia Geographic Information System is a fully functional PC GIS, data analysis and modelling system (Shoemaker *et al.*, 1997) that simulates, amongst others, nonpoint source load of sediments, nutrients and animal wastes.

BASINS – The Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) is a multipurpose environmental analysis system developed by the USEPA. The system integrates water quality and quantity, landuse, point and nonpoint source loadings with point and nonpoint source models. The nonpoint source model is based on the HSPF model and simulates bacteria along with other constituents such as nutrients, sediment and toxics (<http://www.epa.gov/docs/ostwater/BASINS/>).

5. MITIGATION MEASURES

The overall goal of pathogen management in agriculture is to prevent or reduce pathogen movement into surface waters and to facilitate pathogen die-off.

From a catchment perspective, any control measure that reduces runoff and erosion from agricultural areas will reduce the transport of pathogens to surface waterbodies. Several soil and water conservation practices will have a positive impact on reducing pathogen loads (Rosen, 2000).

Interventions are generally targeted at four control points (Figure 5.1).

1. **Import to the farm** – Measures to control pathogen import to the farm is intended to prevent the initial infection of farm animals by these pathogens.
2. **Animal operation amplification/proliferation** – measures to break the cycle of pathogen amplification or proliferation in the intensive animal feeding unit.
3. **Waste management** – measures targeted at the storage of waste products and management of different waste streams on the farm, and
4. **Export from the farm** – measures to constrain the export of pathogens from the farm in surface water streams and rivers.

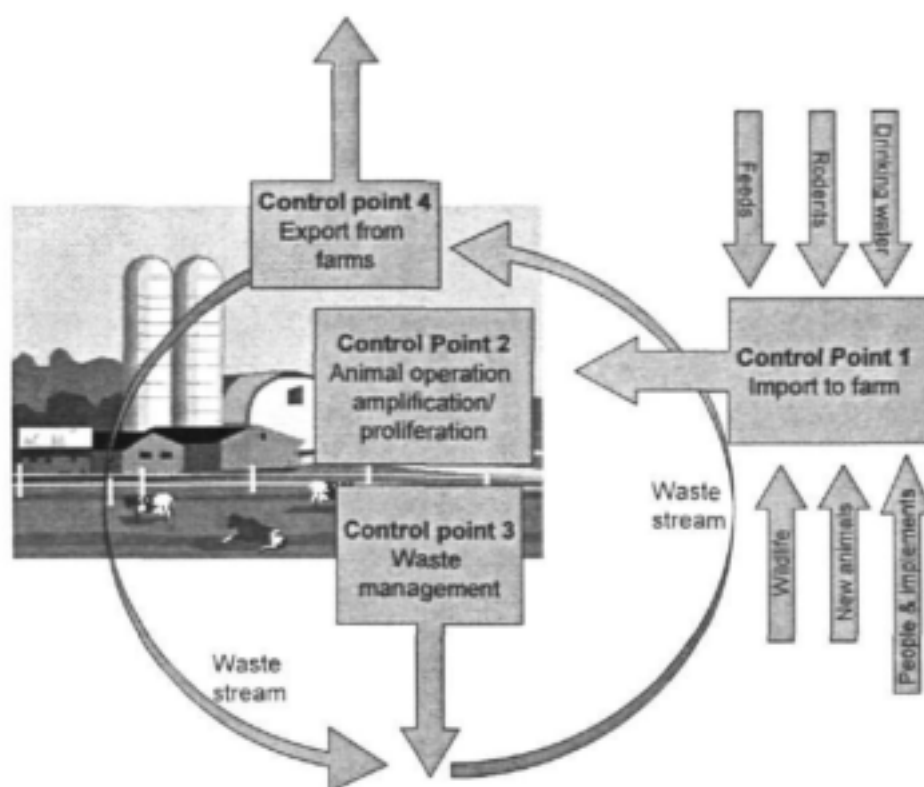


Figure 5.1 Multi-barrier points for pathogen control in agricultural operations (USDA, 2000)

Rosen (2000) identified a number of conservation practices that are targeted at different control points and are aimed at reducing pathogen export from a farm (Table 5.1).

Table 5.1 Selected conservation practices that can reduce pathogen loading directly or indirectly to a catchment

Practice	Import control (source) Amplification/proliferation	Waste management	Export control (transport)
Composting facility	Indirect	Direct	Indirect
Constructed wetland		Indirect	Direct
Filter strips and grassed waterways			Direct
Residue management		Indirect	Direct
Riparian buffers			Direct
Nutrient management		Direct	
Sediment basins			Direct
Waste management systems	Direct	Direct	
Irrigation water management			Direct
Prescribed grazing including exclusions and fencing		Direct	Direct

The Natural Resource Conservation Service of the US Department of Agriculture developed a series of National Conservation Practice Standards (NRSC, 2003). Each conservation practice standard has information on why and where the practice is applied and the minimum quality criteria that must be met during the application of the practice for it to achieve its intended purpose. An example of how these practices are applied to the management of pathogens is illustrated in Figure 5.2. The name of the practice is listed as well as the code number in the National Handbook of Conservation Practices.

A similar knowledge base of soil and water conservation practices is available on WOCAT (World Overview of Conservation Approaches and Technologies). WOCAT was established as a global network of soil and water conservation specialists and it is organised as a consortium of national and international institutions. WOCAT (www.wocat.net) has an online searchable knowledge base of technologies and approaches that can help specialists make informed decisions at a field scale and at a planning level. For pathogen management, a specialist can search for a portfolio conservation practices that would be appropriate for a specific region.

According to Van Coller (pers. comm.) of the Directorate: Water Use and Irrigation Development (Department of Agriculture), a similar catalogue of conservation practices do not exist for South Africa although some guidelines exist for the management of soil erosion on their web site (www.agis.agric.za). They often refer to overseas literature and databases such as WOCAT when providing guidance to farmers and farming organisations.

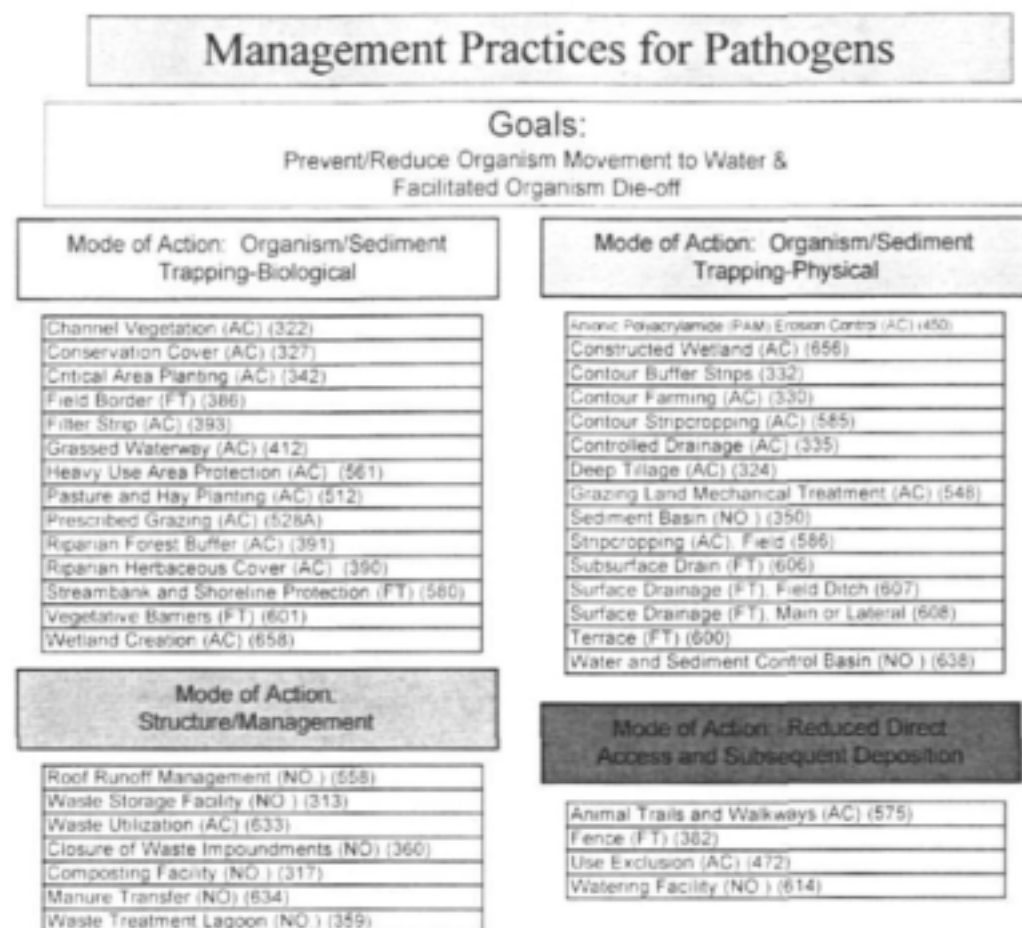


Figure 5.2 An extract of conservation practices described in the USDA National Handbook of Conservation Practices that can be applied to the management of pathogens at a farm scale

5.1 CONTROL OF PATHOGEN IMPORT TO FARMS

The first point of control is to prevent animals from getting infected in the first place with pathogens. This is generally achieved by focussing on a number of sources (Rosen, 2000):

- **Animal feeds** – The primary pathway for pathogens to access animals is through the feeding process and measures are normally taken to prevent contamination of animal feeds. Rodents that contaminate grain are often a source of *Cryptosporidium* and *Salmonella* infection to young calves.
- **Drinking water** – Drinking water of animals can be a source of pathogens especially when the drinking water is contaminated with manure. Measures are generally taken to protect drinking water sources.
- **New livestock** – New livestock contaminated with pathogens can infect healthy animals. New animals are generally separated or quarantined to prevent potential transmission of pathogens to healthy animals.
- **People and implements** - Visitors and the transfer of implements between farms is a potential pathway of pathogen transmission. Implementing biosecurity measures such as cleaning boots, animal handling equipment and vehicle tyres, generally prevents farm-to-farm transmissions.

5.2 CONTROL OF PATHOGEN AMPLIFICATION AND PROLIFERATION ON FARMS

The aim of this control is to prevent pathogens from spreading within the farm. The maintenance of a healthy animal population is a primary goal for farmers and they take care to reduce the growth and spread of infection on the farm. Common practices include the frequent cleaning of stables and animal housing units, cleaning of equipment and foot traffic, securing animal feed against contamination by other animals, humans or pets.

The management of waste on a farm is of particular importance to prevent the spread of infection. Bacteria can proliferate under the right conditions but protozoa, helminths and viruses generally do not multiply outside the host animal. Several options are used to treat manure and these include aerobic lagoons, anaerobic lagoons, composting, and constructed wetlands. If properly implemented to manage waste streams, these measures can control pathogen numbers and the spread of pathogens on a farm. Rosen (2000) describes these control measures as well as control of land applications of manure in greater detail.

5.3 CONTROL OF PATHOGEN EXPORT FROM FARMS

The final control point of control is aimed at preventing the export of pathogens from a farm. This includes control over diseased or contaminated animals, management of pastures, and physical barriers such as buffers and filter strips. Nutrient management strategies to reduce runoff and erosion would have a positive impact on preventing the export of pathogens from a farm. Some measures are specifically aimed at the management of *Cryptosporidium* and *Giardia*. In this case the focus of control is on young calves less than 3 months old, specifically the handling of the calves and their manure.

In South Africa, the WRC developed guidelines for the handling of manure from intensive animal feeding units to control pollution from feedlots, piggeries etc (Funke *et al.*, 1984) and these would be effective for the prevention of on-farm proliferation of pathogens and for preventing the export of pathogens from farms.

6. INFORMATION GAPS AND RESEARCH NEEDS

6.1 DISCUSSION

The monitoring of waterborne pathogens are difficult and in the case of protozoa like *Giardia* and *Cryptosporidium*, very expensive. Agencies undertaking monitoring therefore rely to a large degree on monitoring organisms that indicate pollution from faecal sources. Monitoring is further complicated because water samples need to reach the analysing laboratory within 24 hours of collection. The result is that rivers in rural areas where the impacts of farming would be most evident are often poorly monitored for microbial water quality. Modelling pathogen loadings and the fate in rivers and reservoirs is therefore difficult due to very limited observed data.

There is a wide range of models that can be used to simulate the fate of pathogens in surface waters at various scales. The author is of the opinion that greatest short to medium term need is to find typical pathogen loading rates for South African hydrological conditions and farming practices. The focus should be on the generation and transport of pathogens from a farm to a river. Once discharged into a river, the fate of pathogens appears to be better understood and existing river and reservoir models can adequately describe how pathogens are transported and die off.

6.2 RESEARCH NEEDS

A number of research and information needs have been identified:

- Quantifying the extent of the problem in South Africa – microbial monitoring systems have in general been designed to characterise the impact of densely populated, poorly serviced settlements or of point sources such as wastewater treatment plants effluents. The spatial extent of agricultural sources of high pathogen loads appears to be largely unknown. There is a need to identify particular high-risk agricultural management practices and assess the extent to which these occur in specific catchments, contribute to pathogen loads in surface waters and, ultimately, result in waterborne disease outbreaks. Such an investigation should also include a survey of the prevalence of diseases in livestock that can impact on the human population.
- Pathogen export parameters - There is a need to derive typical pathogen export parameters for South African conditions and farming practices.
- The ecology of waterborne pathogens – There is a need for understanding the ecology of waterborne pathogens from source, viability and survivability from the host through the entire water system.
- Impacts on riparian zones – There is a need to investigate the impacts of livestock grazing in riparian areas and the association with the outbreak of waterborne diseases in humans using those water sources for drinking water purposes. This includes the identification of practices to reduce the impact of livestock in riparian areas on waterborne pathogens.
- Wildlife and waterborne diseases – There is a need to investigate the impact of wildlife on waterborne diseases and impact this may have on rural communities and eco-tourism.
- Source identification – There is a need to identify of the sources of waterborne pathogens through pathogen “finger printing” techniques and their significance in the outbreak of waterborne diseases.

Hazard identification – There is a need to quantify the concentrations of waterborne pathogens that are necessary to create an outbreak of waterborne diseases in a human population.

7. REFERENCES

- Brown, L C and Barnwell, T O. 1987. *The enhanced stream water quality models QUAL2E and QUAL2E-UNCAS: Documentation and user manual*. Report no. EPA/600/3-87/007. US Environmental Protection Agency.
- Bruce-Grey-Owen Sound Health Unit. 2000. *The investigative report of the Walkerton outbreak of waterborne gastroenteritis May – June, 2000*. [Online].
Available: <http://water.sesep.drexel.edu/outbreaks/walkerton.htm>
- Cassell, E A, Meals, D W, Aschmann, S G, Anderson, D P, Rosen, B H, Kort, R L and Dorioz, J M. 2002. *Use of simulation mass balance modelling to estimate phosphorus and bacteria dynamics in watersheds*. *Water Science and Technology*, 45(9): 157-168.
- Department of National Health and Population Development. 1978. *Guide: Permissible utilisation and disposal of treated sewage effluent* [Online].
Available: http://www.dwaf.gov.za/Dir_WQM/docs/Pol_PermisUtilisation.doc
- Du Preez, M and Gericke, M. 1998. *Protozoan parasites in source and drinking water used by semi-urban and rural communities*. Proceedings of the Biennial conference of the Water Institute of Southern Africa (Volume 2), 4-7 May 1998, Cape Town.
- Department of Water Affairs and Forestry, South Africa. 1998. *IMPAQ – User manual for the decision support system*. Amatole Water Resources System Analysis. Phase 2. Department of Water Affairs and Forestry, Pretoria.
- Funke, J W, Knoesen, J G and Venter, J C. 1984. *Guide for the Handling of Manure from Intensive Animal Feeding Units*. Water Research Commission
- Harding, W R and Paxton, B R. 2001. *Cyanobacteria in South Africa: a review*. WRC Report No TT 153/01. Water Research Commission, Pretoria.
- Hipsey, M. 2002. *Three-dimensional modelling of Cryptosporidium fate and transport in Myponga Reservoir, South Australia*. *CWR Models: Bytes and Nybbles* 7:3-4
- Jarmey-Swan, C, Bailey, I W, Howgrave-Graham, A R. 2001. Ubiquity of the water-borne pathogens, *Cryptosporidium* and *Giardia*, in KwaZulu-Natal populations. *Water SA*, 27(1): 57-64
- Jenkins, A, Kirkby, M J, McDonald, A and Naden, P. 1984. A process based model of faecal bacterial levels in upland catchments. *Water Science and Technology*, 16 (5-7) 453-462.
- Menge, J G, Haarhoff, J, König, E, Mertens, R and van der Merwe, B. 2001. Occurrence and removal of *Giardia* and *Cryptosporidium* at the Goreangab Reclamation Plant. *Water Science & Technology: Water Supply*. Vol 1(1): 97-106
- Neelakantan, T R, Brion, G M and Lingireddy, S. 2001. Neural network modelling of *Cryptosporidium* and *Giardia* concentrations in the Delaware River, USA. *Water Science and Technology*, Vol 43(12): 125-132
- Nelson, H. 1997. *The contamination of organic produce by human pathogens in animal manures*. Available online: eap.mcgill.ca/_private/vl_head.html

Newman, R F, Hooper, T D, Powell, G W and Njenga, F M. 2003. *The influence of range practices on waterborne disease organisms in surface water of British Columbia. A problem analysis.* Res. Br., B.C. Min. For., Victoria, British Columbia, Technical Report 008.
Available online: www.for.gov.bc.ca/hfd/pubs/docs/tr/tr008.htm

NRSC. 2003. *National Handbook of Conservation Practices. National Conservation Practice Standards* [Online]. Natural Resources Conservation Services, US Department of Agriculture.
Available: http://www.ftw.nrcs.usda.gov/nhcp_2.html

Rosen, B H. 2000. *Waterborne Pathogens in Agricultural Watersheds.* Water Science Institute, US Department of Agriculture.
Available online: <http://www.wsi.nrcs.usda.gov/products/waterborn-pathogens.html>

Said, H M. 2003. *Prevalence of Cryptosporidium species in young calves at Pretoria Experimental Farm and Irene Dairy Farm.* Unpublished report. Department of Microbiology and Plant Pathology, University of Pretoria.

Salter, H E, Schoombie, S W and Pretorius, E. 2003a. A review of faecal coliform modelling in surface waters I: New and traditional approaches. (Submitted to Water SA for publication).

Salter, H E, Schoombie, S W, Gericke, O J and Truscott, M. 2003b. A review of faecal coliform modelling in surface waters II: River water quality models. (Submitted to Water SA for publication).

Shoemaker, L, Lahlou, M, Bryer, M, Kumar, D and Kratt, K. 1997. *Compendium of tools for watershed assessment and TMDL development.* Report No. EPA841-B-97-006, US Environmental Protection Agency.

United States Department of Agriculture. 2000. Waterborne pathogen information sheets: *Cryptosporidium* and *Giardia*, and *Escherichia coli* O157:H7. USDA Water Science Institute.

Van Collier, A. 2003. Personal communication. Directorate: Water Use and Irrigation Development, Department of Agriculture.

Van Ginkel, C E, Hohls, B C, Belcher, A, Vermaak, E and Gerber, A. 2000. *Assessment of the Trophic Status Project.* Internal Report No. N/0000/00/DEQ/1799. Institute for Water Quality Studies, Department of Water Affairs and Forestry, Pretoria.

Van Niekerk, H. 2000. *National Microbial Water Quality Monitoring Programme: First report on the identification and prioritisation of areas in South Africa with a potentially high health risk due to faecally polluted surface water.*
Available online: www.dwaf.gov.za/iwqs/micrbiol/document/prioritisefin2.htm

Venter, S N, Kühn, A L and Harris, J. 1998. A method for the prioritisation of areas experiencing microbial pollution of surface water. *Water Science and Technology*, 28(12): 23-27

Venter, S N, Steynberg, M C, du Plessis, G, de Wet, C M E, Hohls, D, Rodda, N and Kfir, R. 1996. *Tools for microbial water quality assessment of South African Rivers.* WRC Report No 380/1/96. Water Research Commission, Pretoria.

DISCUSSION PAPER 5:

**ECONOMIC NON-POINT SOURCE POLLUTION
CONTROL POLICY**

by

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APPENDICES

APPENDIX A : Mathematical conditions for cost effective non-point source pollution control policy design1

ECONOMIC NON-POINT SOURCE POLLUTION CONTROL POLICY

The purpose of this paper is to give a short overview of the important factors that should be taken into account when designing non-point source (NPS) pollution control policy. Although policy design is dependent on knowledge of the integral linkages between agricultural production and damage from water quality changes the processes governing NPS pollution is not discussed in this paper. See the other sections of this report for an overview of these processes.

1. ECONOMICALLY EFFICIENT CONTROL OF AGRICULTURAL NON-POINT SOURCE POLLUTION

Griffen and Bromley (1982:548) distinguish the economic problem of agricultural pollution into three separate categories. First, the sediment, nutrients, and chemicals removed by runoff represent a loss of resources to the individual farmers. These costs are borne privately by farmers. Second, a temporal externality exist if the discount rate of the individual farmer is greater than the social discount rate, or if the farmer has a planning horizon which is shorter than society's, the farmer will mine the soil resource at a faster rate than is socially optimal. Third, physical resources lost by the individual farm must appear elsewhere in the environment. In sufficient quantities, these resources are pollutants. Since water is the primary transport media for these resources, water pollution by soil, nutrients and chemicals is an agricultural externality. More specifically, it is classified as a spatial externality emphasising the location of pollution. Although agricultural runoff generates two kinds of externalities (temporal and spatial), only the spatial externality is considered in this review. Spatial externalities exist because farmers have no incentive to consider the cost pollution imposes on others and therefore pollution occurs at unduly high levels (Baumol and Oates, 1988 cited in by Qiu, 1996). More specifically agricultural non-point source (NPS) pollution can be classified as technological diseconomy (negative externality) since it causes a misallocation of resources and benefit conflict between the polluters and pollution victims (Qiu, 1996).

Economic theory suggests several ways to design policies that provide the appropriate incentives for farmers to account for the costs of their pollution (Ribaudo, Horan, and Smith, 1999). To obtain an efficient solution one has to maximise expected net economic benefits which is calculated as the private net benefits of production (aggregate farm profits) minus the expected economic cost of pollution (Ribaudo *et al.*, 1999). Since it is impossible to accurately predict damages due to the varying nature of pollutant runoff and transport decisions it can only be based on the expectation of what damages will be.

More formally economic efficiency is governed by the following three conditions (Ribaudo *et al.*, 1999):

- a) *For each input and each site, the marginal net private benefits from the use of the input on the site should equal the expected marginal external damages from the use of the input.* Thus, the extra private net benefit gained by using more of a specific input at a specific site should be equal to the increased damage associated with using the input. When farmers ignore external damages the condition will be violated resulting in the use of pollution-causing inputs that are too high.
- b) *A site should be brought into production as long as profits on this site are larger than the resulting expected increase in external damage.* This condition defines the optimal amount of land in production. By implication, the amount of land expected to produce profitably is greater than

optimal when external damages are ignored and as a result runoff contributing to ambient pollution levels are too high.

- c) *Technologies should be adopted on each site such that the incremental impact of each technology (relative to the next best alternative) on expected social net benefits is greater than or equal to the incremental impact on expected damages.*

From the above efficiency conditions, it is clear that the equimarginal principal is the driving force to achieve efficient allocation of resources. Since water quality damages needs to be taken into account the tree efficiency conditions represents economic tradeoffs involving farm profitability and water quality. Quantifying these tradeoffs are a prerequisite for meaningful policy to control NPS pollution (Johnson, Adam and Perry, 1991).

Estimating net social benefits maybe troublesome because no market for water quality exists and therefore the benefit function from improving water quality is rarely available if not at all. Several non-market valuation techniques have developed over the years of which contingent valuation, travel cost and hedonic property values are used most commonly (Willis, 1993). Application of these nonmarket valuation techniques in South Africa is scarce with the exception of Oellermann and Darroch (1994), who quantified the total willingness to pay for the perseverance of wetlands. However, Tietenberg (2000) caution the use of non-market valuation techniques stating that while these methods may provide estimates that are good enough to indicate that the benefits from water quality improvements are large and worth pursuing, it may not be reliable enough to use in choosing an economic efficient level of pollution. Using the correct benefits is especially important because pollution abatement costs are high.

The conclusion is that a policy goal with the aim to maximise expected net economic benefits is unachievable since the relationship between economic damage and NPS pollution is largely unquantified.

2. POLICY GOALS AND COST-EFFECTIVE CONTROL OF AGRICULTURAL NON-POINT SOURCE POLLUTION

In the absence of reliable benefit estimates a method commonly used to provide policy makers with useful information is to conduct cost effectiveness analysis (Tietenberg, 2000). Environmental policies are cost-effective if they achieve some measurable objectives or goals at least cost. The process of designing comprehensive policies for controlling NPS pollution therefore consists of defining appropriate policy goals, choosing appropriate instruments and setting these instruments at levels that will achieve the goals at least cost (Ribaudo *et al.*, 1999).

Due to the problems associated with quantifying economic damage from pollution, reducing expected damages from pollution, although desirable does not constitute a measurable policy goal. Instead, it is necessary to adopt alternative goals that are measurable and that are believed to reduce expected damages (Horan and Ribaudo, 1999; Ribaudo *et al.*, 1999). Potential alternatives include physical (i.e., ambient water quality) or production related measures (i.e., input use).

The National Water Act (Act 36 of 1998) provides for both resource-directed and source-directed measures to control NPS pollution (DWAF, 1998). Resource directed measures aim at providing an appropriate level of protection for different water resources through the use of a resource classification system. As part of the resource classification system resource quality objectives must be established for each water resource. These objectives might prove useful to establish measurable policy goals for cost effective analysis. However, it is important to note that ambient water quality goals such as those used for point sources may not be feasible for NPS pollution control policies because of the stochastic nature of NPS pollution. This is mainly due to the fact that the probability distributions associated with damages and ambient water quality are not the same and control of one distribution does not necessarily imply control of the other (Horan and Ribaudo, 1999). When environmental outcomes are stochastic pollution control strategies should be aimed at improving the distribution of outcomes rather than some scalar value (McSweeney and Shortle, 1990).

When goals are based on production related activities policy makers are provided with a more direct means of reducing expected damage. These goals are advantages because they are deterministic and it is easy to verify whether the goals are met or not. Care should, however, be taken to choose goals such that it affects producer choices mostly correlated with pollution and that the pollution increasing substitution effects are limited or of no consequences (Horan and Ribaudo, 1999).

3. FACTORS AFFECTING COST-EFFECTIVE NON-POINT SOURCE POLLUTION CONTROL IMPLEMENTATION WHEN USING DIFFERENT BASES

After measurable NPS policy goals are identified a logical next step in pollution control is to set policy instruments at levels that will achieve the goals at least cost. The emphasis of this section is not on alternative instruments itself but rather on the bases to which they are applied since it has a bearing on the performance of the chosen instrument. Furthermore, in the presence of the usual caveats concerning the omission of income effects, transactions costs and time the specified emission incentives or standards and the management incentives or standards are equally allocatively efficient in achieving a specific emission target (Griffen and Bromley, 1982). Instruments can be applied either to farmers' actions (design based) or to the results of their actions (performance based). Appendix A contains the mathematical conditions for alternative cost-effective NPS pollution control policies.

3.1 PERFORMANCE BASED

Performance bases refer to pollution emissions generated by farmers' production decisions or the resulting ambient water quality.

When controlling NPS pollution under heterogeneous conditions through performance-based (emissions or ambient water quality) standards, a separate standard needs to apply to each farmer to achieve the water quality objective at minimum cost. On the other hand, a uniform performance-based incentive will be allocatively efficient when applied to heterogeneous pollution sources since the marginal cost of abatement are equalised across the pollution sources yielding the least cost method of achieving the specified pollution target. A necessary condition for this to happen is that the marginal damages from each source should be constant regardless of the source of the extra pollution (Griffen and Bromley, 1982; Segerson, 1988). However, marginal damages from different pollution sources might vary, as in the case of emissions entering a river at different places with different assimilative capacities (Helfand and House, 1995) or where the pollution emissions are reduced by vegetative buffer strips (Qiu, 1996). Qiu (1996) clearly demonstrates that one needs differentiated incentives/standards when some farms are likely to contribute more heavily to marginal ambient water pollution than others. By implication, this means that two farms with the same production and pollution emission generating potential will need differentiated economic incentives if they contribute differently to water quality problems. From the above discussion it is clear that pollution-based discriminatory standards, taxation or subsidisation, which may not be socially acceptable, will occur when farms contribute differently to pollution (Griffen and Bromley, 1982; Segerson, 1988).

Given the fact that policies with the goal of pollution emission control are infeasible due to prohibitively expensive monitoring (Shortle and Dunn, 1986) ambient water quality will be the preferred performance-base. Ambient water quality based instruments are advantageous for the following reasons (Ribaudo *et al.*, 1999). First, economic theory suggests instrument bases (ambient pollution levels) should be close to the externality (damages from pollution). Second, ambient pollution can be monitored without the resource management agency having to observe the actions of each producer. Thirdly, producers have flexibility in their production and pollution control decisions under ambient-based incentives in that they may utilise any private knowledge they may have to further reduce costs, or they may alter their decisions as economic and environmental conditions change. However, these advantages may soon be disappearing when concerning the other complexities of the instrument base.

Ambient water quality measures the resulting water quality from a group of farmers and therefore farmers within this group are not subsidised or taxed according to their own performance but on that of

the group (Segerson, 1988). Ambient-based instruments may prove to be complex for the farmers since the cost effectiveness of such a policy will be determined by each farmer's ability to evaluate how their actions and the actions of others affect the incentive base (Ribaudo *et al.*, 1999). The information requirements will thus be very high. Furthermore, free riding may be a problem when policy instruments are used to control pollution from a group of farmers. An important feature of an ambient-based policy is that it shifts the information burden from the regulators to the farmers. Cabe and Herriges (1992) demonstrate that ambient-based instruments may not achieve the desired outcome if farmers' beliefs regarding the fate and transport of pollutants are different from those of the regulators. The desirability of the policy may strongly hinge on the regulator's ability to ascertain farmers' beliefs about pollutant fate and transport, or to influence these beliefs through education or extension (Cabe and Herriges, 1992).

3.2 DESIGN-BASED

Controlling management practices that leads to pollution was proposed in literature as a practical alternative to controlling pollution itself since management practices are observable whereas emissions are difficult or prohibitively expensive to observe. Design-based instruments refer to instruments applied to producer's variable input use and production technology choices. Next section distinguishes between two different types of designed based instruments namely expected runoff bases and instruments targeted more directly on input and technology choices.

3.2.1 Input and technology based

Shortle and Dunn (1986) extended Griffen and Bromley's analysis (Griffen and Bromley, 1982) given in Appendix A by incorporating the stochastic nature of pollution generation. Furthermore, they assumed that the farmers have better information regarding the effects of changing management practices on farm profits when reallocating resources in response to policy than the regulators when choosing a policy. Their general conclusion was that incentives on management practices (which may be inputs) would generally outperform quantity controls. However, cost-effective control may be very restrictive since differentiated instruments (economic incentives or standards) needs to apply to each management practice on each farm even though marginal pollution damages may be constant across farms, as long as the farms are heterogeneous. Therefore, input-based, cost-effective solutions to NPS pollution depends on having detailed information on input usage and on the ability to regulate all inputs contributing to pollution simultaneously (Larson, Helfand and House, 1996). By implication, each farmer's inputs need to be priced or subsidised individually in the case of economic incentives and consequently discriminatory taxes are always the case if one is concerned with cost effective solutions. Another problem arises when those farmers facing lower taxes buy large quantities and resell to those farmers who face high taxes (Helfand and House, 1995).

The conclusion is made that cost-effective control of NPS pollution is difficult in the presence of spatial variability of the natural resource base, weather and the site specific nature of NPS pollution resulting in detailed information needs regarding input use, technology and management practices.

3.2.2 Expected runoff

Expected runoff is classified as designed based because one may use information on input and technology use with NPS pollution models to estimate expected pollution loads (Ribaudo *et al.*, 1999). Expected runoff is of concern because it has some of the same desirable characteristics of performance based instruments mentioned above.

To apply expected runoff-based instruments effectively producers needs information on how their production and pollution control decisions influence expected runoff. This information may be supplied

to the farmers by the NPS pollution control enforcing agency for each field for alternative input and technology choices. Since the schedule is based on the controlling agency's expectations about runoff, farmers' expectations of the pollution process will not lead to any inefficiencies such as in the case of performance based instruments.

However, using a NPS pollution simulation model to develop these schedules will be expensive since the model needs to be run for each site and alternative input and technology uses. Furthermore, compliance can only be established if the input and technology use of each of the farmers on each of the sites are monitored adding to monitoring and administrative cost. Also, producers may be reluctant to truthfully provide any private information to the resource management agency for fear that this information might be used against them in the design of environmental policy (Ribaudo *et al.*, 1999).

3.3 CONCLUSIONS

From this section it is clear that the implementation of cost-effective policies are difficult because of the difficulty to observe pollution emissions, free riding, information needs on management practices, different perceptions of the control agency and farmers regarding the expected impact of pollution control activities, number of controlling entities to achieve cost effectiveness and the fact that discriminatory taxes might be socially unacceptable. A key issue in determining economic efficient pollution control is the ability to account for spatially distributed site-specific information. Instruments that ignore spatial variability such as uniform instruments will only achieve cost effectiveness under the rare situation where marginal damages from different pollution sources are equal.

Although implementation of cost-effective policies are in most of the cases not practically feasible they still provide us with benchmarks from which to guide more practical alternative policies especially in the absence of data on transaction cost.

4. SECOND-BEST ALTERNATIVE POLICIES

Cost effective and efficient outcomes define the optimal actions taken to satisfy NPS pollution control policy goals in an ideal world where the set of policy instruments is not restricted and there is no transaction costs associated with implementing optimally designed policies (Ribaudo *et al.*, 1999). However, transaction costs and policy limitations are important and should not be ignored when designing policies. In practice, with these limitations and costs, the best possible outcome would achieve policy goals at the lowest cost, given the types of instruments that are used and the costs associated with using them. Such an outcome is generally referred to as second-best (Ribaudo *et al.*, 1999). While second-best policies are optimal in practice, their economic performance in the sense of being able to achieve a goal at least cost is still measured relative to the ideal of a cost-effective or efficient baseline. This provides a useful method of comparison between alternative types of policies, especially when data on transactions costs are unavailable (as they often are).

Several researchers have evaluated uniform NPS pollution control policies (Carpentier, Bosch and Batie, 1998; Larson, Helfand and House, 1996; Qiu, 1996; Helfand and House, 1995; Shortle and Dunn, 1986). Helfand and House (1995) compared four relatively easy implementable alternative second-best policies to control nitrate leaching from two soils used for lettuce production. These include a) uniform input tax applied to all pollution sources; b) uniform reduction in inputs contributing to pollution based on a specific percentage reduction for all sources; c) uniform tax on single inputs contributing to pollution and d) uniform input restrictions. Uniform NPS pollution policies are not cost effective because the same set of instruments applies to all farmers under heterogeneous conditions. These researchers found little difference between the cost-effective solution and the alternative second-best uniform policies even when applied to heterogeneous conditions. However, generalisation of the results is not possible since it is dependent on the degree of heterogeneity taken into account. Most researchers show that the cost effectiveness of alternative policies to control NPS pollution increases as the level of heterogeneity taken into account increases when ignoring transactions cost.

Controlling single inputs may not lead to satisfactory results due to substitution effects that take place between input use (Ribaudo *et al.*, 1999). Controlling one input may trigger the use of another input that may increase the pollution problem. Furthermore, input use is dependent on the type of technology adopted. Thus, the cost effectiveness of second best alternative policies can be improved if technology adoption was also controlled.

5. TARGETTING

An innovative approach to increase the efficiency of second best policies to achieve cost-effective outcomes is targeting of pollution reduction responsibilities. Given the high cost of pollution control, targeting is an attractive option to NPS pollution control since it improves the cost effectiveness of pollution control expenditures by governments and it can allocate pollution reduction responsibilities to areas in the catchment having the lowest control cost (Bosch and Wolfe, 1998). Dickinson, Rudra and Wall (1990) questioned the effectiveness and efficiency of a targeting approach to control NPS pollution in all situations reasoning that the desirability and necessity of targeting is primarily a function of the spatial variability of factors relating to pollutant loads. In fact, spatial variability in terms of resource base, geographical location and weather can be identified as the rationale for using targeting to improve cost effectiveness since differences in these factors are likely to result in varying pollution loadings and marginal abatement costs. Empirical evidence shows that when ignoring the spatial variability in a catchment the estimated pollution levels and pollution reduction costs will be biased if pollution is correlated to agricultural productivity (Opaluch and Segerson, 1991). Furthermore, ignoring the spatial variability in the catchment may cause the researcher to overlook: 1) pollution control opportunities available to the farmers thereby possibly overestimating the control cost and 2) opportunities to reduce cost by targeting areas with low cost of pollution control (Bosch, Pease and Stone, 1998).

Various criteria such as productivity index (Runge, Larson and Roloff, 1986) sediment delivery rate, sediment delivery ratio, slope and distance to the nearest water source (Fox, Umali and Dickinson, 1995) have been evaluated for targeting purposes. However, it is still unclear what criteria should be used since the same criteria may not be as effective as the other when applied to different catchments emphasising the site-specific nature of the problem. Empirical evidence has shown that targeting strategies based on economic criteria are more cost effective than targeting strategies based on physical or pollution related criteria such as sediment delivery ratios (Fox, Umali and Dickinson, 1995; Ribaud, 1989). Thus, the ability of a targeting scheme to increase the cost effectiveness of programs to reduce agro-pollution depends on two aspects: a) how well the targeting scheme approximates the marginal cost of pollution reduction and b) the total area (hectares/farms/subcatchments) affected (Carpentier, 1996). Using these criteria Carpentier, Bosch and Batie (1998) showed that targeting based on spatial information reduced compliance cost by nearly 80% when compared to the uniform standard that ignores spatial variability.

It is concluded from this section that detailed information about the spatial variability of factors influencing pollution loads may hold promise to further reduce compliance cost by targeting the pollution reduction responsibilities to those having the lowest control cost.

6. INDIRECT INSTRUMENTS FOR CONTROLLING NON-POINT SOURCE POLLUTION

Because pollution is a process whereby pollutants are transported through various pathways and mediums, controlling factors that influences these pathways or the transport medium will necessarily have an impact on pollution levels. Lee and Howitt (1996) suggest that on-farm water conservation offer great potential to improve water quality through reduced drainage. Weinberg, Kling and Wilen (1993) included on-farm water conservation strategies when evaluating the impacts of water markets on water quality externalities and concluded that a water market provides the necessary economic incentives to conserve water resulting in water quality improvements through reduced drainage.

Given fully allocated water supplies, water can be conserved from agriculture and made available for sale in three ways (Weinberg, Kling and Wilen, 1993): a) water applications can be reduced *ceteris paribus*, potentially reducing crop yields due to water stress; b) irrigation system performance can be improved to maintain crop yields as water applications are reduced; or, c) cropping patterns can be changed. However, these strategies may not always proof to reduce drainage. If land is available and farmers are allowed to expand there irrigation operation drainage may increase during some periods of the growing season (Oosthuizen and Grové, 2001). Furthermore, while reduced drainage may lead to reduced pollution emissions, reduced drainage can also reduce the volume of water in rivers and hence its dilution capacity (Connor and Perry, 1999). When taking dilution into account Connor and Perry (1999) showed that negative groundwater externalities may result from water trade even when institutional protection for return flow externalities is provided.

7. CONCLUSIONS

Due to the problems associated with implementing performance based instruments design based proofs to hold most promise to control NPS pollution. No single instrument will be the most effective in all situations because of spatial variability and the site specific nature of NPS pollution. The tool or combination of tools best suited for a particular problem is an empirical issue based on policy goals, local conditions, and the costs of acquiring information. The first step in formulating effective NPS pollution control policy is to gain insight into the unique factors affecting NPS pollution through interdisciplinary research and information gathering. Only after sufficient information on the pollution problem and alternatives are gathered will one be able to generate economic environmental tradeoffs necessary for efficient NPS pollution control policy.

8. REFERENCES

- Baumol, W.J. and Oates, W.E. 1988. *The Theory of Environmental Policy*, second edition, Cambridge University Press.
- Bosch, D.J. and Wolfe, M.L. 1998. Soil and Water Quality in Agriculture, Assessing and Improving. In R.A. Meyers (ed). *Encyclopedia of environmental analysis and remediation*. New York: John Wiley & Sons.
- Bosch, D.J., Pease, J. and Stone, N. 1998. Incorporating Spatial Variability in Economic Analyses of Agricultural Nonpoint Source Pollution Control. *Paper presented at the Annual Meeting of the Southern Information Exchange Group 70, Economics and management of risk in agricultural resources*, 26-28 March, 1998, Gulf Shores, Alabama.
- Cabe, R. and Herriges, J.A. 1992. The Regulation of Non-Point-Source Pollution under Imperfect and Asymmetric Information. *Journal of Environmental Economics and Management*, 22:134-146.
- Carpenter, C.L., Bosch, D.J. and Batie, S.S. 1998. Using Spatial Information to Reduce Costs of Controlling Agricultural Nonpoint Source Pollution. *Agricultural and Resource Economics Review*, 27(1): 72-84.
- Carpenter, C.L. 1996. Value of Information for Targeting Agro-Pollution Control: A Case Study of the Lower Susquehanna Watershed. Ph.D thesis. Department of Agricultural and Applied Economics, Virginia Polytechnic Institute and State University, Blacksburg.
- Connor, J.D. and Perry, G.M. 1999. Analyzing the potential for water quality externalities as the result of market water transfers. *Water Resources Research*, 35(9):2833-2839.
- Dickinson, W.T., Rudra, R.P. and Wall, G.J. 1990. Targeting Remedial Measures to Control Non-point Source Pollution. *Water Resources Bulletin*, 26(3):499-507.
- Fox, G., Umali, G. and Dickinson, T. 1995. An Economic Analysis of Targeting Soil Conservation Measures with Respect to Off-site Water Quality. *Canadian Journal of Agricultural Economics*, 43(1995):105-118.
- Griffen, R.C. and Bromley, D.W. 1982. Agricultural Runoff as a Non-point Externality: A Theoretical Development. *American Journal of Agricultural Economics*, August 1982:547-552.
- Helfand, G.E. and House, B.W. 1995. Regulating Non-point Source Pollution Under Heterogeneous Conditions. *American Journal of Agricultural Economics*, 77:1024-1032.
- Horan, R.D. and Ribaudo, M.O. 1999. Policy Objectives and Economic Incentives for Controlling Agricultural Sources of Non-point Source Pollution. *Journal of the American Water Resources Association*, 35(5):1023-1035.
- Horan, R.D., Shortle, J.S. and Abler, D.G. 1998. Ambient Taxes When Polluters Have Multiple Choices. *Journal of Environmental Economics and Management*, 36:186-199.
- Larson, D.M., Helfand, G.E. and House, B.W. 1996. Second-Best Tax Policies to Reduce Non-point Source Pollution. *American Journal of Agricultural Economics*, 78:1108-1117.
- Lee, D.J. and Howitt, R.E. 1996. Modeling Regional Agricultural Production and Salinity Control Alternatives for Water Quality Policy Analysis. *American Journal of Agricultural Economics*, 78:41-53.

- McSweeney, W.T. and Shortle, J.S. 1990. Probabilistic Cost Effectiveness in Agricultural Non-point Pollution Control. *Southern Journal of Agricultural Economics*, July 1990:95-104.
- Oellermann, R.G. and Darroch, M.A.G. 1994. Estimating wetland preservation values: a Wakkerstroom case study. *Agrekon*, 33(4):202-205.
- Oosthuizen, L.K. and Grove, B. 2001. Optimale bestuur van wisselvallige waterbesikbaarheid op plaas- en streeksvlak met inagneming van risiko en die omgewing. WRC report No 645/1/01. Pretoria: The Water Research Commission.
- Opaluch, J.J. and Segerskon, K. 1991. Aggregate Analysis of Site-Specific Pollution Problems: The Case of Groundwater Contamination from Agriculture. *Northeastern Journal of Agricultural Economics*, 20(1):83-97.
- Johnson, S.L., Adams, R.M. and Perry, G.P. 1991. The on-farm costs of reducing groundwater pollution. *American Journal of Agricultural Economics*, 73(4): 1063- 1073.
- Qiu, Z. 1996. Integrated Assessment of Agricultural Non-point Source Pollution in Goodwater Creek Watershed, Missouri. Ph.D thesis. Faculty of the Graduate School, University of Missouri-Columbia.
- Ribaudo, M.O. 1989. Targeting the Conservation Reserve Control Program to Maximize Water Quality Benefits. *Land Economics*, 65(4):320-332.
- Ribaudo, M.O., Horan, R.D. and Smith, M.E. 1999. Economics of Water Quality Protection from Non-point Sources: Theory and Practice. *Agricultural Economics Report No 782*. Resource Economics Division, Economic Research service, US Department of Agriculture, Washington DC, USA.
- Runge, C.F., Larson, W.E. and Roloff. 1986. Using productivity measures to target conservation programs: a comparative analysis. *Journal of Soil and Water Conservation*, 41:45-49.
- Segerson, K. 1988. Uncertainty and Incentives for Non-point Pollution Control. *Journal of Environmental Economics and Management*, 15:87-98.
- Shortle, J.S. and Dunn, J.W. 1986. The Relative Efficiency of Agricultural Source Water Pollution Control Policies. *American Journal of Agricultural Economics*, August 1986:668-677.
- Department of Water Affairs and Forestry, South Africa. 1998. National Water Act. Act No 36 of 1998. 20 August 1998.
- Tietenberg, T. 2002. *Environmental and Natural Resource Economics*. Addison-Wesley Publishers. 656pp.
- Weinberg, M., Kling, C.L. and Wilen, J.E. 1993. Water Markets and Water Quality. *American Journal of Agricultural Economics*, 75:278-291.
- Willis, D.B. 1993. Modelling economic effects of stochastic water supply and demand on minimum stream flow requirements. Unpublished Ph.D.-thesis. Department of Agricultural Economics, Washington State University, Pullman.

APPENDIX A

**Mathematical Conditions for Cost Effective Non-point
Source Pollution Control Policy Design**

APPENDIX A

MATHEMATICAL CONDITIONS FOR COST EFFECTIVE NON-POINT SOURCE POLLUTION CONTROL POLICY DESIGN

Pricing in the form of taxes and subsidies and standards are often used to control externalities, both of which may attain the same pollution reduction if appropriately specified. However, the base to which it is applied has far reaching implications when it comes to the implementation of these policies. In this section the theoretical foundation of incentives and standards when applied to different bases are summarised starting with a point source and extending it to NPS. The implementation of these policies is discussed in the following section.

A1 POINT SOURCE EMISSIONS BASED

If the polluters are profit maximisers and there is perfect competition, the pricing and standard method can be presented using profit maximisation framework, as shown by Griffin and Bromley (1982) which is summarised by (Qiu, 1996) as follows.

Suppose there are J farms which have their own production activities which generate emissions. Let y^j be the production bundle of farm j with y_n^j being the n^{th} element (positive or negative) of that vector. Positive activities represent outputs and negative ones are inputs. There are N goods or activities, i.e., $n = 1, \dots, N$. Emissions by farm j are non-negative and are denoted z^j . Total emissions from all the farms are limited to Z^* . Farm j 's production set is given implicitly by $f^j(y^j, z^j) \leq 0$. These relationships become equalities when the farms fully exploit their production abilities, which is assumed, here.

Given these specifications, society's problem is to:

$$\text{Maximise: } \sum_{j=1}^J p y^j \quad (1.1)$$

$$\text{subject to: } f^j(y^j, z^j) \leq 0 \quad \text{for all } j \quad (1.2)$$

$$\sum_{j=1}^J z^j \leq Z^* \quad (1.3)$$

where:

p price vector ($1 \times N$)

Society's Lagrangian is:

$$L = \sum_{j=1}^J p y^j - \sum_{j=1}^J \alpha^j f^j(y^j, z^j) - \mu \left(\sum_{j=1}^J z^j - Z^* \right) \quad (1.4)$$

where:

α^j and μ are appropriate Lagrange multipliers

Assuming that the implicit production functions are concave and that all constraints are binding, the following first-order conditions (FOCs) describe the optimal choice of production activities:

$$p_n - \alpha^j f_n^j = 0 \quad \text{for all } j \text{ and } n \quad (1.5)$$

$$-\mu - \alpha^j f_z^j = 0 \quad \text{for all } j \quad (1.6)$$

Subscripts on functionals denote partial derivatives and those on vectors denote particular vector elements.

Consider the private side. Without consideration of the externality, the farm j 's problem is to maximise profit (py^j) subject to the constraints on production, $f^j(y^j, z^j) \leq 0$. The FOC for this problem is equation (1.5).

The above problem can be modified by introducing an economic incentive for reducing emissions. Let s represent the per unit incentive on emissions and Z be the incentive base level, a predetermined quantity from which greater emissions are charged at rate s and lesser emissions are subsidised at the same rate. Farm j 's problem is now:

$$\text{Maximise} \quad py^j + s(Z - z^j) \quad (1.7)$$

$$\text{subject to:} \quad f^j(y^j, z^j) \leq 0 \quad \text{for all } j \quad (1.8)$$

The Lagrangian function is:

$$L^j = py^j + s(Z - z^j) - \delta^j f^j(y^j, z^j) \quad (1.9)$$

Optimality conditions are derived from the following FOCs:

$$p_n - \delta^j f_n^j = 0 \quad \text{for } n \quad (1.10)$$

$$-s - \delta^j f_z^j = 0 \quad (1.11)$$

Comparing equations (1.5) and (1.6) to equations (1.10) and (1.11) shows that the social optimum (not necessarily being Pareto optimum) can be realised by choosing an economic incentive which equals the shadow price of the environmental constraint, i.e., $s = \mu$. In this case, the optimal incentive, s^* , is the same for all farms and is dependent upon the incentive base level, Z . Later, it is shown that the optimal incentive may vary across farms when different farms make different contributions to the externality.

An equally efficient set of regulations can be mathematically derived using profit functions. Let $\pi^j(p, s)$ represent farm j 's optimal profit function, which is specified as a function of the price vector and the incentive level:

$$\pi^j(p, s) = \text{Max} \quad py^j + s(Z - z^j) \quad (1.12)$$

$$\text{subject to:} \quad f^j(y^j, z^j) \leq 0 \quad (1.13)$$

Applying Hotelling's lemma, the farm's optimal output of pollution is the derivative of the profit function with respect to the cost of emitting the pollutant. Evaluating this partial derivative at the appropriate prices and the optimal incentive, we have an equally optimal emission standard for farm j .

$$z^j = z^j(p, s^*) = \frac{\partial \pi^j(p, s)}{\partial s} \bigg|_{p, s^*} \quad \text{for each } j \quad (1.14)$$

Equation (1.14) describes a set of optimal emission standards which, when enforced, will achieve the targeted aggregate emissions such as Z at least-cost. The emission standard defined by equation (1.14) are as efficient as the least-cost incentive, $s^* = \mu$. Each of the policies is the dual of the other. Due to this dual relationship the allocative efficiency of both programs is guaranteed to be the same (Griffin and Bromley, 1982).

A2 NON-POINT SOURCE EMISSION GOAL

Implementation of the two policies discussed above requires that the pollutant be monitorable. In the case of NPS pollution, monitoring of Non-point source emissions are either infeasible or impractical. However, the inputs and outputs of production activities are monitorable. As shown by Griffin and Bromley (1982), the least-cost framework developed in the previous section can be extended to develop some environmental policies which target the inputs and outputs associated with NPS pollution. They extended the framework by introducing an emission generating function.

Returning to the specification of the previous model, assume that every farm is fully utilising its productive abilities, i.e., $f^j(y^j, z^j) = 0$ for all j . Applying the implicit function theorem, there exists a neighbourhood about y^j and smooth function g^j such that $f^j[y^j, g^j(y^j)] = 0$ for all j throughout the neighbourhood. The only restrictive assumption necessary to apply this theorem is $f_z^j \neq 0$. In the presence of unabated pollution, each farm chooses $f_z^j = 0$. However, environmental policy is intended to direct the farm away from this point and therefore the assumption is valid. The implication of this theorem is that an emission generating function is expressible as a continuously differentiable function of all inputs and outputs. The formulation is completely general, accommodating inputs and/or output determinants of NPS pollution. For convenience, assume that the emission generating function, $g^j(y^j)$, does not differ among farms, so the superscripts on this function can be dropped.

Based on this functional relationship, Griffin and Bromley proposed four least-cost environmental policies to control NPS pollution. The first is a non-point incentive, which is equivalent to the incentive formulation given in the previous section and denoted as s^* . The second is a system of non-point standards, which is still expressed by equation (1.14). The difference between point and non-point incentives and standards is not based on the theoretical formulation but on the operation and enforcement. Non-point incentives and standards can be implemented through monitoring the farm agricultural activities and then calculating the emissions using the emission generating function $g^j(y^j)$.

The third is called management incentives. Instead of a tax or subsidy based on emissions, the management incentives are imposed on inputs and outputs of agricultural activities. Let σ denote the vector of incentives attached to the elements of y^j and σ_n is the incentive on activity y_n^j . Let Y represent the management incentive base levels. Each Y_n in Y is a predetermined and arbitrary quantity from which greater activity levels are charged and lesser activity levels are subsidised at the rate σ_n . The private problem for farm j can be formulated as follow:

$$\text{Maximising} \quad py^j + \sigma(Y - y^j) \quad (1.15)$$

$$\text{subject to} \quad f^j[y^j, g^j(y^j)] = 0 \quad (1.16)$$

The private Lagrangian is and FOC are as follows:

$$L^j = py^j + \sigma(Y - y^j) - \delta^j f^j[y^j, g(y^j)] \quad (1.17)$$

$$p_n - \sigma_n - \delta^j (f_n^j + f_n^j g_n) = 0 \quad \text{for all } n \quad (1.18)$$

Society's problem is slightly different because of the removal of z^j as an independent variable. The social Lagrangian and FOC are specified as:

$$L = \sum_{j=1}^J py^j - \sum_{j=1}^J \alpha^j f^j[y^j, g(y^j)] - \mu \left[\sum_{j=1}^J g(y^j) - Z^* \right] \quad (1.19)$$

$$p_n - \mu g_n - \alpha^j (f_n^j + f_n^j g_n) = 0 \quad \text{for all } n \text{ and } j \quad (1.20)$$

Comparison of (1.18) and (1.20) identifies the least-cost management incentives as:

$$\sigma_n^* = \mu g_n = s_n^* g_n \quad \text{for all } n \quad (1.21)$$

This system of incentives is not necessarily the same for all farms because derivatives of the emission generating function are evaluated at different activity levels.

The fourth environmental policy is management practices, which is the dual of management incentives. Let $\pi^j(p, \sigma)$ denote farm j 's profit function in the presence of management incentives. Hotelling's lemma can be used to specify the optimal levels on individual agricultural activities. These optimal activity levels are called management practices, and are defined as:

$$y_n^{j*} = y_n^j(p, \sigma^*) = \left| \frac{\partial \pi^j(p, \sigma)}{\partial \sigma_n} \right|_{p, \sigma^*} \quad \text{for all } n \text{ and } j \quad (1.22)$$

The management practices will induce the least-cost achievement of the environmental emission standards.

A3 WATER QUALITY GOAL

Ambient-based instruments have the advantage that it shifts the location of monitoring from the production choices (technology, input use, crops etc.) of farms that are suspected of contributing to environmental degradation to the environmental media itself (Horan *et al.*, 1998).

Segerson (1988) was the first to propose the use of an ambient-based tax to control NPS pollution. Her research shows that a uniform tax based on ambient water quality can achieve an efficient level of NPS pollution in a catchment with multiple polluters only when the marginal benefits of abating pollution are constant. Horan, Shortle and Abler (1998) extended Segerson's analysis by exploring the design of uniform ambient-based taxes when each firm has a multiple choice set. Although the polluters pay an uniform marginal tax rate per unit of additional ambient pollution the correct marginal incentives are maintained since the polluters do not pay the same rate per unit of pollution abatement

(Segerson, 1988). The latter depends on each polluter's contribution to marginal ambient water quality levels.

Qiu (1996) used the concept that each polluter (catchment) contributes differently to marginal water quality problems to derive a conceptual framework with variable economic incentives that allows for the implementation of the tax at sub-catchment level. Following Qiu (1996) the least-cost framework with variable economic incentives can be developed by introducing the concept of pollution contribution rate.

Suppose the objective of society is to limit the aggregate pollution to Z^* . Aggregate pollution is a continuous and differentiable function of emissions generated by all farms, $h(z)$, where z is a vector of emissions generated by farms. The constraint on the relationship between emission at all farms and aggregate pollution is:

$$h(z) \leq Z^* \quad (1.23)$$

Instead of using $\sum_{j=1}^J z^j \leq Z^*$, society's problem can be reformulated using constraint (1.23). The social Lagrangian becomes

$$L = \sum_{j=1}^J p y^j - \sum_{j=1}^J \alpha^j f^j(y^j, z^j) - \mu [h(z) - Z^*] \quad (1.24)$$

The FOCs are:

$$p_n - \alpha^j f_n^j = 0 \quad \text{for all } n \text{ and } j \quad (1.25)$$

$$-\mu h_z^j - \alpha^j f_z^j = 0 \quad \text{for all } j \quad (1.26)$$

where:

h_z^j partial derivative of the contribution function, $h(z)$, with respect to z^j and interpreted as farm j 's pollution contribution rate.

By comparing equations (1.25) and (1.26) to equations (1.5) and (1.6), the optimal economic incentive for farm j with minimum costs is shown to be:

$$s_j^* = \mu^* h_z^j$$

The optimal economic incentives vary across farms when farms have different contributions to aggregate pollution. Similarly, a system of standards on emissions from all farms can be derived as the dual of the variable economic incentive using the same framework presented above.

**TERMS OF REFERENCE FOR THE LONG-TERM
RESEARCH PROGRAMME TO PREDICT
AGRICULTURAL NON-POINT SOURCE POLLUTION
FROM FIELD TO CATCHMENT SCALES FOR
SELECTED AGRICULTURAL NPS POLLUTANTS**

by

AHM Görgens

Sigma Beta

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TERMS OF REFERENCE FOR A LONG-TERM RESEARCH PROJECT

1. INTRODUCTION

The overall objective of the project was to involve major players active in the field of non-point source pollution in agriculture, in a scoping exercise and knowledge gap analysis to develop the terms of reference for a longer term project that would establish an integrated model for the prediction of agricultural non-point source (NPS) pollution from field to catchment scales for the major agricultural NPS pollutants. This section describes the terms of reference and how it was compiled.

The terms of reference was broadly guided by three sets of key research questions generated by the team during a brainstorming workshop in August 2003:

- What are the impacts of non-point sources in agriculture on water quality pollution? How significant is non-point source pollution from Agriculture? How good are first order estimates? Can we predict future WQ problems/ do we have emerging WQ problems? What are the time and space lag issues? Can we identify and quantify individual contributors to catchment scale water quality problems?
- Will controlling/managing non-point sources in agriculture improve fitness for use? What impacts do landuse changes have on water quality? How do we manage non-point source pollution? If agriculture is a big source, are farmers aware of it and do they have the tools to manage/address it? What implementation tools are feasible? What is the cost effectiveness of mitigation measures/tools? How do you quantify economic externalities?
- Do we have monitoring at various scales to support NPS control and management?

1.1 INTRODUCTION TO THE FAIM KNOWLEDGE GENERATION HIERARCHY

The terms of reference (TOR) for the long-term research project has been structured according to the "FAIM Knowledge Generation Hierarchy" (Görgens & van Wilgen, 2004). The FAIM Hierarchy is based on the understanding that the ultimate aim of most research initiatives in the domains of water resources, land resources and ecological services is to provide support to decision-making regarding the management of such resources. The quality of support provided to such management decision-making is dependent on sound linkages with relevant fundamental and applied knowledge about biophysical processes and on the integration of biophysical with other forms of knowledge. These requisite linkages are the keys to an *interactive, overlapping* hierarchical structure of knowledge and the layers of research, methodologies and tools which feed knowledge-generation. For convenience, this structure has been called the "FAIM" (Fundamental / Applied / Integrative / Management) knowledge-generation hierarchy, and is outlined in Figure 1.

The project is structured into four overlapping and iterative phases, where each phase corresponds roughly with one of the levels in the Hierarchy.



Figure 1 Diagram of the four overlapping and iterative phases of the FAIM Knowledge Generation Hierarchy.

1.2 OUTPUTS FOCUS

The TOR is broadly guided by the Project Objectives and the aforementioned Key Research Questions, but the detail and the scope of the requisite individual research tasks, will depend on the incremental Outputs required from each Phase of the Project. For that reason, the TOR has been focused on these incremental Outputs, rather than on the research tasks. Research Proposals submitted in response to this TOR are expected to unpack the tasks required by each Output.

1.3 FOCUS AND GUIDING CRITERIA

The following aspects provided the focus and guiding criteria for the TOR:

- The focus of the TOR is on priority pollutants, nutrients (nitrogen and phosphorus), sediment, and pesticides.
- Salinity, metals and pathogens were not included.
- The focus should, as far as possible, be on using existing models and tools.
- There was a realisation that new field- and catchment-scale observation and monitoring studies could not be avoided.
- Pesticide studies would be undertaken only to field-scale.
- It was not practicable to fully integrate models across all spatial scales.
- Streamflow and groundwater: relative positions in terms of scale.
- Economics of NPS control measures in agriculture should be considered.

2. TERMS OF REFERENCE FOR THE LONG-TERM RESEARCH PROGRAMME TO PREDICT AGRICULTURAL NON-POINT SOURCE POLLUTION FROM FIELD TO CATCHMENT SCALES FOR SELECTED AGRICULTURAL NPS POLLUTANTS

TERMS OF REFERENCE

KEY STRATEGIC AREA: WATER UTILISATION IN AGRICULTURE

Thrust 4: Water Resource Protection and Reclamation in Agriculture

Programme 2: Impact Assessment and Environmental Management of Agricultural Production

Title: Development of an integrated modelling approach to prediction of agricultural non-point source (NPS) pollution from field to catchment scales for selected agricultural NPS pollutants.

Objectives

General:

Developing an integrated modelling approach to prediction of agricultural NPS pollution from field- to catchment-scale for selected NPS pollutants.

Specific:

1. Establishment of fate-of-NPS-pollutant and other requisite data sets at field-, laboratory-, and quaternary catchment-scale, suitable for model improvement, development and verification.
2. Improvement of understanding of on-farm NPS pollution control measures and their modelling requirements.
3. Establishment of an improved field-scale model for simulation of agricultural NPS pollution loadings for phosphorus, nitrogen, selected pesticides and sediments, as well as for simulating the beneficial impacts on nearby receiving waters of on-farm NPS pollution control measures.
4. Establishment of an improved quaternary catchment-scale model for simulation of agricultural NPS pollution loadings for phosphorus, nitrogen and sediments, as well as for simulating the downstream benefits of on-farm NPS pollution control measures at the larger scale.
5. Development of guidelines about model usage to examine economic trade-offs and feasibility of agricultural NPS pollution control measures at field- and catchment-scale.

Rationale

It is increasingly recognised that non-point source (NPS) (or: diffuse) pollution plays a major role in the degradation of water quality; specifically with respect to salinity, eutrophication (nutrient enrichment), sediments, pathogens, pesticides and some heavy metals. It is furthermore, increasingly accepted that it is infeasible to properly manage water quality without addressing the contribution from non-point sources. Consequently, attention is increasingly devoted to the quantification of NPS pollution and to identify means to control it cost-effectively at source.

Since most of the land area is utilised for agricultural activities, agriculture has both locally and internationally been implicated as a major source of NPS pollution. It is thus necessary to assess the contribution that the different agricultural activities make to the different manifestations of NPS pollution, to devise the means through which these can be controlled and to determine and predict the effect that control measures will have to reduce NPS pollution.

During 2001 the WRC published the document, "A guide to non-point source assessment" (report TT 142/01, by GC Pegram and AHM Görgens), which aims to assist users in understanding their NPS assessment needs in a particular situation, and thereby to enable them to identify and apply appropriate NPS assessment techniques. This Guide development process was generic and did not include any direct model development and testing, nor did it deal with the agricultural sector specifically. Understanding the production, delivery, transport and use components of agriculture-derived NPS loadings of water resources and having a predictive ability about the fate of agriculture-related NPS constituents are discrete research themes that would enhance the usefulness of that Guide, in the agricultural domain.

The contribution that irrigation and dry-land cultivation activities makes towards the salinisation of water resources has been studied and funded by the WRC for quite some time and is currently still receiving attention. Other water quality issues of concern that are potentially aggravated by agricultural activities are eutrophication (through fertiliser leaching and wash-off from human settlements), sediments (as a result of erosion), pathogens (from intensive animal production units), pesticides (through the application of insecticides, fungicides and herbicides) and some heavy metals. Although agricultural activities that give rise to the latter water quality issues have been the subject of previous studies, our present level of knowledge concerning them, and, particularly, our ability to model the fate of these non-conservative constituents in the hydrological cycle, are not as advanced as for salinisation of water resources.

Cognisant of this situation, the WRC initiated a one-year Scoping Study in April 2003 to develop a Knowledge Review, with a focus on the following agriculture-derived NPS pollutants: phosphorus, nitrogen, pesticides, sediments, heavy metals and pathogens. The Scoping Study concluded with the following three deliverables:

- Knowledge Review of Modelling Non-Point Source Pollution in Agriculture from Field to Catchment Scale
- First Order Estimate of the Contribution of Agriculture to Non-Point Source Pollution in Three South African Catchments: Salinity, Nitrogen and Phosphorus
- These Terms of Reference.

The "Knowledge Review" and "First Order Estimate" documents are available from the WRC. These two documents provide essential background information for prospective tenderers for this Project.

The multi-disciplinary nature of the planned research, as well as its extensive scope make it essential that prospective tenderers should consider forming consortia to maximise the knowledge, skill and experience that they may be able to offer. Proposals to undertake only particular parts of the TOR will not be considered.

Deliverables

Phase One: Fundamental research outputs: nitrogen, phosphorus, sediment and selected pesticides

- 1.1: Report on cross-referenced description and prioritisation of primary NPS processes for production of each of N, P, sediment and selected pesticides from agricultural activities.
- 1.2: Report on comparative theoretical evaluation of adequacy of the algorithms for these primary NPS processes in existing field-scale and catchment-scale water balance and water quality models identified in the Knowledge Review as promising candidates for use in South Africa. Tenderers must indicate their preferences, but the final choice of models will be made during

the inception stage of the project, and will depend on track record, software code availability and scientific support.

- 1.3: Annual and final reports on continuous data sets of all priority processes relevant to each pollutant type, assembled in experimental plot-scale or field-scale fate-of-pollutant studies, including wash-off zone, root-zone, vadose zone and groundwater observations, as appropriate. The experimental units should be in two different bioclimatic regions of South Africa and should distinguish between dry-land (rain-fed) and irrigated examples of agriculture, typical to the region involved. The experimental units should, as far as possible, be situated in the two nested gauged quaternary catchments referred to in the Phase Three description below.
- 1.4: Interim and final reports on a comparative critique of aforementioned models via fate-of-pollutant verification studies in typical soil profiles selected in the experimental plot or field settings, and supported by laboratory-based pollutant transport and leaching tests on selected soil columns.
- 1.5: Report on identification of and motivation for selection of the most promising field-scale model for each pollutant type for South African applications in the agricultural NPS field and a description of further development needs of the identified models. The model development recommendations of the Knowledge Review must be closely considered in the formulation of this output.

Phase Two: *Applied and predictive research outputs – nitrogen, phosphorus, sediment and selected pesticides*

- 2.1: Report on the development of a field-scale model for N, P and sediments. The model algorithms for the fate of the individual pollutants can be either integrated or closely-coupled. The model development recommendations of the Knowledge Review must be closely considered in the formulation of this output.
- 2.2: Interim and final reports on a configured and verified field-scale model for N, P and sediments developed under output 2.1, using the continuous data sets provided in output 1.3.
- 2.3: Report on the development of an integrated or closely-coupled field-scale model for selected pesticides and sediments. The model development recommendations of the Knowledge Review must be closely considered in the formulation of this output. The water balance "engine" of this model should correspond closely with that produced for N, P and sediments under output 2.1.
- 2.4: Interim and final reports on configured and verified field-scale models for the selected pesticides, using the continuous data sets provided in output 1.3.
- 2.5: Report on a prioritised set of on-farm NPS pollutant control measures, based on existing knowledge, and descriptions of additional field-scale process modelling requirements and other functional improvements dictated by the need to model on-farm pollutant control measures.
- 2.6: Reports on development of an improved field-scale model for, respectively, the selected nutrient and pesticide pollutant groups, taking into account the development needs described in output 1.5, the improvements dictated by output 2.5 and the model development recommendations of the Knowledge Review.

Phase Three: *Integrative and interface tools research: nitrogen, phosphorus and sediment (not pesticides)*

- 3.1: Annual and final reports on continuous data sets of pollutant concentrations and loadings in streamflow and groundwater, as well as all relevant geobioclimatic and land-use data, for two sets of nested gauged quaternary-size catchments (100 – 500km²) in two different bioclimatic regions of South Africa. The catchments should represent significant agricultural land-uses

- and the nested gauged sub-catchments should be at an area-scale that is about two orders of magnitude finer than the full-sized catchment.
- 3.2: Reports on configured versions of the catchment-scale models listed in output 1.2, based on the continuous data sets provided in output 3.1.
 - 3.3: Interim and final reports on a comparative critique of the configured catchment-scale models via fate-of-pollutant calibration and verification studies, based on the nested catchment data sets provided in output 3.1. If the time-frame of the project is too brief for the assembly of adequate observed pollutant data sets, then the critique must be based on a heuristic comparison of model outputs and model component states, given identical model inputs.
 - 3.4: Report on the identification of and motivation for selection of the most promising catchment-scale model for South African applications in the agricultural NPS field, and a description of further development needs of the identified model.
 - 3.5: Report on development and verification of an improved catchment-scale model for local use, taking into account the development and improvement needs formulated under outputs 1.5, 2.5 and 3.3.
 - 3.6: Report on guidelines for the use of the field-scale models to help parameterise field-scale processes in the catchment-scale model.
 - 3.7: Report on development and testing of a graphical user interface (GUI) to provide a transparent environment for interchange of information between the selected field-scale model and the catchment-scale model, as well as between the current time-series management and GIS support tools of the respective existing models.

Phase Four: Management support – nitrogen, phosphorus and sediment

- 4.1: Report on a model-based case study, by use of the field-scale models, of the relative beneficial impacts at fine scales, of alternative on-farm NPS control measures on pollutant delivery to receiving streamflow and groundwater. This is a desk-study and does not include a field demonstration of these beneficial impacts.
- 4.2: Report on a case-study, based on the two nested catchments selected under output 3.1, demonstrating downstream pollution mitigation by alternative individual or combined on-farm NPS control measures.
- 4.3: Report on a case study, based on the details of the two nested catchments selected under output 3.1, conceptually demonstrating an appropriate methodology for evaluation of the economic trade-offs of agricultural NPS pollution control measures. This methodology should make appropriate provision for the potential economic influence of waste discharge charges.
- 4.4: Report on a case study, based on outputs 4.1, 4.2 and 4.3, quantifying the economic trade-offs of agricultural NPS pollution control measures, both at farm-scale and at catchment-scale.
- 4.5: Report on guidelines for the use of the field- and catchment-scale models configured and improved in phases 1, 2 and 3 for evaluation of the economic trade-offs of agricultural NPS pollution control measures.

Time Frame

4 years, from 1 July 2004 to 30 June 2008

Total Funds Available

Approximately R2 500 000, with annual budgets as follows:

<i>Financial year</i>	<i>Approximate Amount</i>
2004/2005	R500 000
2005/2006	R650 000
2006/2007	R650 000
2007/2008	R500 000
2008/2009	R200 000

3. REFERENCES

Görgens, A H M and Van Wilgen, B W. 2004. Invasive alien plants and water resources in South Africa: current understanding, predictive ability and research challenges. *South African Journal of Science*, 100 (Jan/Feb): 27-33.

Other related WRC reports available:

Policy considerations for non-point source management in South Africa

GC Pegram • AHM Görgens • GE Quibell

Classification of nonpoint sources is required for legal purposes, because the appropriate management approach varies for different nonpoint source types. Although all human activity potentially has an impact on the receiving water environment, cost-effective nonpoint source management requires the prioritisation of those activities with the greatest impact. Therefore, a simple classification system is required to enable and focus nonpoint source control and enforcement. "Cutoff levels" should define the intensity or size of these activities, above which nonpoint source management is required, based on the potential impact on the receiving water environment.

Classification of nonpoint sources should be based on source type (land use activity), with cutoff levels reflecting the size and intensity of the activity which has a potential impact on the receiving water environment.

The listing of those activities which are classified above the "cutoff levels", should be enabled through the water act, as the basis for nonpoint source management. Regulations for control and enforcement should be specified according to these classes (i.e. listed activities).

Report Number: 696/1/98

ISBN: 1 86845 496 7

Also see the following report:

Development of a guide to assess non-point source pollution of surface water resource in South Africa

Quibell G; Pegram GC; Moolman J; Matji MP; Hohls B, Görgens AHM

Report Number 696/2/03

ISBN: 1 86845 991 8

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