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Development of Indicators of Sustainability for Effluent Reuse in the Intensive Livestock Industries: Piggeries and Cattle Feedlots

Resource Manual

FLOT.318B
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FEEDLOTS

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Australian Pork Limited, NSW Environment Protection Authority and Meat and Livestock Australia.

Glossary of Terms

Acid soil - Soil with a pH less than 7

ALFA - Australian Lotfeeders Association

Alkaline soil - Soil with a pH exceeding 7.

ANZEEC - Australian and New Zealand Environment and Conservation Council

APL - Australian Pork Limited

Aquifer - A water-bearing rock formation able to transmit significant quantities of water to a bore, spring or watercourse

ASAE - American Society of Agricultural Engineers

AUD - Australian dollars

Available nutrient - That portion of any element in the soil that can be readily absorbed and assimilated by growing plants.

BeefBal - Nutrient Mass Balance Spreadsheet for Feedlots

BOD - Biological Oxygen Demand – a measure of the quantity of oxygen used by micro-organism in the biochemical oxidation of organic matter in a period of 5 days under specific conditions

CEC - Cation Exchange Capacity – That total of exchangeable cations that a soil can adsorb.

COD - Chemical Oxygen Demand

CRC - Cooperative Research Centre

Dispersion - Disintegration of micro-aggregates into clay, silt and sand grains.

DUAP - Department of Urban Affairs and Planning

Duplex soil - Soil that shows a sharp change in texture between the surface layer and the subsoil. Also called a texture-contrast soil.

EC - Electrical Conductivity

EIS - Environmental Impact Statement

EPA - Environmental Protection Aguthority

Erosion - The wearing away of land surface by rain or wind, removing soil from one point to another e.g. gully, rill or sheet erosion.

FLIAC - Feedlot Industry Advisory Committee

FSA - Feedlot Services Australia

Hard-setting - Occurs when the soil surface “melts” together when wet, drying to a hard and impermeable surface over 10 mm thick.

Infiltration - Downward entry of water into the soil.

IP Act - Integrated Planning Act, 1997

Leaching - Process where soluble nutrients e.g. nitrogen are carried by water down the soil profile.

LBL - Load Based Licensing

LCA - Life Cycle Impact Assessment

MEDLI - Model for Effluent Disposal using Land Irrigation

MINAS - Minerals accounting system

MLA - Meat & Livestock Australia

MWPS - Mid-west Planning Service

NFAS - National Feedlot Accreditation Scheme

NPI - National Pollution Inventory

pH - A measure of the acidity or alkalinity of a product. The pH scale ranges from 1 to 14. A pH of 7 is neutral, a pH below 7 is acidic and a pH above 7 is alkaline.

Pig Bal - Nutrient Mass Balance Spreadsheet for Intensive Piggeries

Pollution - Direct or indirect alteration of the environment causing contamination or degradation.

SAR - Sodium Absorption Ratio – A measure of the sodicity of the water.

SCARM - Standing Committee on Agriculture Resource Management

SCU - Standard Cattle Unit

SEE - Statement of Environmental Effects

Soil solution - The liquid phase of the soil and its solutes (including dissolved nutrients).

Soil structure - The arrangement of primary soil particles into aggregates.

Soil texture - The relative amount of coarse sand, fine sand, silt and clay in the soil.

SPCC - State Pollution Control Commission

SPU - Standard Pig Unit

Surface water - Includes water in dams, lakes, reservoirs, rivers, creeks and all other waterways where rainfall is likely to collect.

TDS - Total Dissolved Solids – The inorganic salts (major ions) and organic matter (nutrients) that are dissolved in water, used as a measure of salinity.

VFC - Victorian Feedlot Code

Waterlogging - Saturation of a soil with water causing displacement of air to the point where there is insufficient oxygen for full root activity.

WMP - Waste Management Plan

WQCQ - Water Quality Council of Queensland

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Issue 1 – Draft (dated 15 June 2002) circulated to the steering committee for comment.

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Issue 4 - Draft (dated 31 March 2003) included updates from the consultation process and was circulated to the steering committee members for comment.

Issue 5 – This Resource Manual and accompanying Summary Document have been compiled to meet the requirements of Australian Pork Limited Consultancy Agreement: 1816 (dated 19 March 2002). It provides information to address the aims and objectives of the Consultancy Agreement as set out by Australian Pork Limited.

This Resource Manual supersedes all previous versions of the document that were distributed for public comment.

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1. INTRODUCTION

Australian Pork Limited (APL), Meat & Livestock Australia (MLA) and NSW Environment Protection Authority (EPA) wish to develop indicators of sustainability for the reuse of effluent and solid by-products from the intensive livestock industries: piggeries and cattle feedlots. These sustainability indicators would demonstrate sustainable effluent and solid by-product reuse for the target industries and could be applied to the EPA-administered load based licensing (LBL) scheme in New South Wales.

The principal aims of this project are to produce a Resource Manual including sustainability indicators for the reuse of effluent and solid by-products from piggeries and cattle feedlots, and tools that producers can use to evaluate and demonstrate their sustainability. Meat processing and rendering, which are undertaken by some operators of piggeries and cattle feedlots, are specifically excluded from this project.

The LBL scheme is New South Wales' pollution licensing scheme. The scheme links licence fees to potential environmental impact. Under the scheme, licensees may receive up to a 100% rebate of load fees for sustainable effluent reuse. The LBL scheme includes piggeries and cattle feedlots under the Intensive Livestock Production activity category. Stage 1 of the LBL scheme has been implemented. This involved a subset of activities that were currently licensed by EPA. The following criteria were considered when defining the initial scope of the scheme:

- The potential for environmental harm.
- The ready availability of load estimation techniques.
- The state of development of the licensing framework for the industry.
- The resources required in implementing the scheme.

The activities to be included in the first phase of the scheme were those with the significant potential for environmental harm, which had traditionally been covered by the licensing scheme. It was originally intended that most of the remaining, currently licensed activities would be progressively phased into the scheme by November 1999.

Load fees for piggeries and cattle feedlots were not included in stage 1 of the LBL scheme. This report provides extensive information on the application of piggery and cattle feedlot effluent and solid by-products. The sustainability indicators that are provided are designed to assist in the sustainable management of reuse systems for these industries and are not suggested as limits to calculate and apply LBL fees.

The authors advise that it is important that NSW EPA and operators of piggeries and cattle feedlots recognise that it is extremely difficult to develop tools for determining and demonstrating sustainability and indicators of sustainability that cover all situations. It is probable that the tools for determining sustainability will overstate the likely risk to the environment in some cases. Consequently, where a significant level of environmental risk or impact is identified, it is critical to confirm that this result is accurate through further investigations.

2. PROJECT OBJECTIVES

This Resource Manual intends to provide readily available data and analysis techniques for evaluating the sustainability of effluent and solid by-product reuse for piggeries and cattle feedlots. It provides suggested sustainability indicators for these intensive livestock industries. Specifically, it:

- Examines the LBL scheme.
- Examines current guidelines and regulatory requirements for piggeries and cattle feedlots.
- Discusses effluent and manure production for piggeries and cattle feedlots.
- Investigates the effects that nutrients and salts can have on soil and water resources.
- Identifies appropriate indicators for sustainable effluent and solid by-products reuse that could be used by piggery and feedlot operators and regulatory agencies to measure environmental performance and improve sustainability via changes in management of the system (whether or not LBL applies).
- Summarises methods for protecting soil, surface and groundwater resources through good design and management.
- Outlines how mass balance principles can be used to decide appropriate nutrient and salt loading rates based on land use. This section includes suggested maximum nutrient application rates based on land use. It also suggests techniques for estimating the loads of key pollutants applied to land by intensive livestock industries licensed by the EPA, and methods for estimating nutrient removal by cropping.
- Defines a risk assessment procedure to be used for deciding the minimum (cost-effective) monitoring requirements that individual facilities could use to demonstrate sustainability. The type and level of monitoring for any facility would depend on the risk to surface water, groundwater and soil resources. This will include suggested monitoring parameters and monitoring frequency for each by-product for reuse (e.g. effluent or solids) and each reuse area at any given enterprise.
- Recommends practices to reduce the risk of adverse environmental impacts from effluent or solid by-products reuse.
- Identifies areas needing further research.

This Resource Manual draws on state, national and international research. Its development has also relied on extensive consultation with those currently undertaking applicable research. It provides the best currently available scientific basis to assess the environmental sustainability of the reuse of effluent and solid by-products from intensive livestock enterprises (piggeries and cattle feedlots). It also provides a useful starting point for the consideration of sustainability indicators for other intensive livestock industries. A separate summary report provides key practical points from the main report e.g. indicators of sustainability, risk assessment and monitoring recommendations.

3. LOAD BASED LICENSING

This section of the document examines the aspects of the NSW LBL scheme that are relevant to intensive livestock industries. It also looks at the overseas application of LBL (or similar schemes) to intensive livestock enterprises.

3.1. Summary of NSW Load Based Licensing Scheme

3.1.1. Background

The Load-Based Licensing Scheme was introduced in July 1999 under the Protection of the Environment Operations (General) Regulation 1998. The LBL Scheme was a major overhaul of the NSW environment protection licensing system.

Previously, licensing fees were mainly based on the scale and type of a licensed activity, or the maximum allowable volume of wastewater permitted to be discharged. The LBL Scheme is based on two key principles:

- The primary measure and limit tool for licensed discharges is the annual pollutant load (or mass emitted) instead of the concentration of pollutants contained in the discharges. This new approach is designed to provide a stronger outcomes-based focus for the licensing system, and thus greater assurance of environment protection. It is also intended to provide greater flexibility for licensees to find cost-effective and innovative options for meeting environmental requirements.
- The pollution load licence fee is designed to provide ongoing incentives for pollutant load reductions. The fee is based on the quantity and type of pollutants discharged, with adjustments for the manner of discharge and the condition of the receiving environment.

The activities to be included in the first phase of the scheme were those with the significant potential for environmental harm, which had traditionally been covered by the licensing scheme. Piggeries and cattle feedlots were not included in Stage 1 of the LBL scheme. This report provides extensive information on sustainability indicators for piggery and cattle feedlot effluent and solid by-product application.

3.1.2. Pollutant Fees

The methodology used to determine the pollutant weighting values was based on internationally acceptable Life Cycle Impact Assessment (LCA) methodology. The broad principles of LCA methodology are set out in ISO10040-Life Cycle Assessment, with detailed steps for impact analysis in the draft ISO14042. Examples of some pollutant fees are provided in Table 1.

TABLE 1 – EXAMPLES OF POLLUTANT FEES UNDER THE LBL SCHEME (NSW EPA, 1998)

Pollutant	$A/tonne Discharged for Enclosed Waters*
Phosphorus (non-marine waters)	7,140
Nitrogen (non-marine waters)	242

*Fees taken from online LBL calculator for sewage treatment.

3.1.3. Load Calculation Protocol

The current version of the Load Calculation Protocol for use by holders of NSW Environment Protection Licences was gazetted on 10 May 2002.

The assessable load of a pollutant is the least of the actual, weighted or agreed load. The actual load of a pollutant is the mass (in kg) of the pollutant released to the environment. The weighted load of a pollutant is the actual load adjusted using specified load-weighting methods that recognise practices or circumstances that effectively reduce the environmental harm without reducing the actual load. The agreed load is a load that will be achieved through future improvements as part of a Load Reduction Agreement.

The methods suggested for calculating actual loads in the Load Calculation Protocol are:

- Source monitoring – this involves directly measuring volume and concentration data either continuously or periodically, for example from an irrigation outlet pipe.
- Emission factor – this uses either generic emission data derived from broad average emission data or site-specific emission factors.
- Mass balance calculations – this assumes that the discharge to the environment is the difference between inputs and outputs. It is only applicable when input and output streams can be accurately quantified. Where the declared error range of the mass balance exceeds 10%, the amount equal to the portion of the error range exceeding 10% must be added to the estimated load values. Mass balance principles can be applied to individual components of an activity or across an entire activity.

Fee reductions of up to 100% for sustainable effluent reuse can be applied. Reuse discount factors for each pollutant are the sum of a 'pollutant management factor' (0, 0.25 or 0.5, where 0 represents sustainable performance) and a 'water management factor' (0, 0.25, 0.5, where 0 represents sustainable performance). Better performance leads to a lower factor and thus greater discounting.

To gain a full discount (0) for nitrogen and phosphorus they must be applied so that they are effectively used for plant growth or sustainable assimilation by the soil system. If nitrogen and phosphorus levels below the plant root zone are rising, the average amount of effluent applied per unit area must be decreased. The sustainable rate of application of nutrients (such as nitrogen and phosphorus) can sometimes limit the quantity of effluent to be used for irrigation in a given area. To obtain the fee discount, licensees must:

- Have developed a 15-year forward management plan that shows how proposed annual nutrient application rates compare with the annual amounts to be taken up by the biological or physical processes of the crop–soil system. This should be done before the construction of the effluent reuse scheme. Nutrient application rates must be based on the sustainable assimilation of nutrients over a rolling 15-year period.
- Review the plan every 3 years to ensure that future planned application rates will continue to achieve sustainable assimilation over a rolling 15-year period.
- Prepare annual nutrient balances showing nutrient application rates and the results of soil monitoring done as set out in the management plan, and how these outcomes compare with those anticipated in the management plan. Documentation of plan and annual balances must be kept for at least 4 years.

To gain a partial discount (0.25) for nitrogen and phosphorus the same criteria apply, except the planning timeframe is only 5-15 years.

A full discount (0.0) for water management is gained if the application rate is controlled by irrigation scheduling or soil moisture monitoring to ensure that effluent or liquid waste does not percolate deeper than the root zone or intersect groundwaters, except during scheduled salt flushing as per management plan.

A partial discount (0.25) for water management is gained if application ceases during and after rainfall as necessary to prevent waterlogging or runoff.

Discount factors for salt management are calculated depending on the TDS concentration (mg/L), the SAR, the concentration of Na^+ and Cl^- (mg/L) and management practices employed. Effluent applied so that nutrient budget requirements are met. The amount of effluent applied is dependent on the value of the above parameters. See http://www.epa.nsw.gov.au/licensing/lblprotocol/index.htm for more information.

3.1.4. Pig Industry in NSW

The latest version of Pigstats (Meo & Cleary, 2000) indicates that NSW is the largest pig producing state in Australia, with approximately 30% of the national herd. There were 922 herds, with a total of 91,000 sows. The nine piggeries with more than 1000 sows comprise over 50% of the state's herd. There are also 16 herds with between 400 and 1000 sows, comprising of another 9,260 sows.

3.1.5. Feedlot Industry in NSW

From the Australian Lotfeeders Association magazine (Lotfeeding – September 2002) the current capacity of NSW feedlots is 310,200 head, with 250,400 head on feed (81% occupancy). This represents about 34% of the Australian industry.

3.2. LBL In Other Countries

3.2.1. Background

There has been very little overseas licensing of intensive livestock industries by "pollutant load". A United Nations Environment Programme summary of schemes from different countries is given at http://www.unep.or.jp/ietc/Publications/TechPublications/TechPub-11/5-4-2.asp. The only relevant scheme is the MINAS system that operates in the Netherlands. Details of this system are summarised below.

3.2.2. MINAS System – Netherlands

The Netherlands, with the highest population density in the world and a large intensive livestock industry, has insufficient land for sustainable reuse of livestock manure. In the past, this resulted in over-application and the subsequent pollution of both land and water. In 1998, the Netherlands reformed their manure and ammonia policy to make farms account for reuse methods, account for differences within sectors and to stimulate technological development and enterprise. The new minerals accounting system (MINAS - http://www.minlnv.nl/international/policy/environ/) involves a registration of the mineral inputs (nitrogen and phosphate) used on a farm in fertilisers and animal feeds, and the mineral output in the form of products and manure. The difference between inputs and outputs is the mineral loss that ends up in the environment. When the loss is larger than the allowable standard, a levy applies.

Loss Standards

The difference between the mineral input and output of the property is the mineral loss. Part of the loss is considered allowable and there is no levy against this portion. In the mid 1990's, average losses in livestock areas in the Netherlands were 65 kg of phosphate and 370 kg of nitrogen per hectare. In 1998, loss standards of 40 kg of phosphate and 300 kg of nitrogen per hectare were introduced (Table 2). The allowable phosphate and nitrogen losses are to be gradually reduced to allow time for adaptation and to avoid unacceptable social and economic consequences. The allowable losses set for 2008/2010 are designed to meet environmental quality objectives. The concern with low allowable phosphate loss standards is addressed below (Exceptions - Reparation dressing).

Levies

Levies are payable when mineral losses exceed the allowable standard. The levy depends on the level of standard exceedance. Farmers pay Dfl 5 ($3.78 AUD - converted 24/5/02) for the first 10 kg of phosphate exceeding the standard and Dfl 20 ($15.10 AUD - converted 24/5/02) for every additional kilogram (Table 2).

To enforce efficient distribution of manure surpluses, the levy for exceeding the phosphate loss standards is relatively high. To be effective, the higher levies needed to exceed the most expensive reuse option. To facilitate the transition to tightened loss standards, the levies made measures such as improved feed, manure application management and manure redistribution at short range worthwhile.

To meet the phosphate loss standards, a producer will most likely meet the nitrogen standards. However, a separate nitrogen levy will be introduced later depending on the cost involved in meeting the standard. It is expected to be relatively low and a progressive system should not be necessary.

Exceptions - Reparation Dressing

Joint studies by the Netherlands government and industries indicate that stricter loss standards can reduce the topsoil phosphate level. In soils with a low phosphate level and in phosphate-fixing soils, the stricter standards can affect soil fertility and reduce crop yields. To counter this effect, an allowance is made for 'reparation dressing'. The allowable phosphate loss can be set at 50 kg per hectare if farmers can prove through soil analysis that phosphate levels are low. The reparation dressing tool is currently being tested for enforceability to enable uniform application.

Non-Intensive Livestock Farms

Farms operating below the livestock density standard can accept manure from more intensive producers. To avoid dumping of manure on properties not regulated under the mineral accounting system, the amount of manure supplied to farms must not exceed the phosphate supply standard (Table 2). The phosphate supply standard, which includes both manure and fertiliser applications, was reduced to 80 kg/ha in 2002.

TABLE 2 – PAST REGULATIONS AND ESTIMATE FOR FUTURE LOSS STANDARDS, PROGRESSIVE LEVIES, SUPPLY STANDARDS, LU THRESHOLDS AND THEIR INTERRELATIONS - NETHERLANDS

	1998	2000	2002	2005	2008 / 2010
Phosphate allowable loss standard (kg P_2O_5/ha)	40	35	30	25	20
Nitrogen allowable loss standard (kg N/ha)(1)	300	275	250	200	180
Low levy (Dfl 5, \$3.78 AUD)(3) for phosphate loss of (kg P_2O_5/ha)	40-50	35-45	30-40	25-30	(2)
High levy (Dfl 20, \$15.10 AUD)(3) for phosphate loss exceeding (kg P_2O_5/ha)	50	45	40	30	(2)
Phosphate supply standard (kg P_2O_5/ha)	-	85	80	80	80
on grassland	120				
on arable land	100				
Registration obligatory at livestock unit (LU) number	2.5	2.5	2.0	2.0	(2)

(1)Standard applies to grassland: exclusive of deposition and mineralization.
(2)To be determined later.
(3)Currency conversions as of May/02.

3.3. Advantages and Disadvantages of LBL for Piggeries and Cattle Feedlots

3.3.1. Advantages of LBL

It is preferable from an environmental sustainability viewpoint to license activities by mass of pollutants released rather than by effluent concentration. A concentration-based approach allows pollutants to be released in low concentrations (i.e ‘the solution to pollution is dilution’). However, the cumulative effect of pollutant releases gives rise to environmental degradation. The concentration-based approach discourages the reuse of water and may encourage water wastage for dilution purposes. This is not as relevant to piggeries or feedlots since the concentration of nutrients and organic matter in the effluent they produce does not meet discharge standards, even after substantial treatment.

For NSW EPA, the main advantage of LBL is the provision of a framework for managing cumulative impacts. It will be much simpler to compare licensees’ performances and impacts based on load information.

3.3.2. Disadvantages of LBL

A major difficulty in applying LBL to effluent and solid by-products reuse areas is determining appropriate indicators of sustainability. It is very difficult to adequately consider the wide variation in natural resources that may exist for an individual enterprise or indeed across the industries and the related utilisation of by-products. Also, piggeries and cattle feedlots primarily reuse their by-products (effluent and solids) in a cropping or pasture system. Cropping and pasture systems will always lose some nutrients to the environment via runoff, leaching or gaseous loss, whether they are fertilised with animal manure by-products, fertilised with inorganic fertilisers or even left in the virgin state (unfertilised). Identifying benchmarks or triggers for assessing the sustainability of a system is very complex due to the large variations in resources, climatic conditions and management practices between sites.

4. Australian Environmental Guidelines & Regulatory Requirements

Each Australian state has a separate regulatory regime for licensing cattle feedlots and piggeries. The regulations can cover both the development phase (development applications) and the operating phase (management plans). A wide range of guidelines provides direction to the regulatory agencies.

This section identifies the current regulatory requirements for effluent and solid by-product reuse in piggeries and feedlots throughout Australia, placing the LBL scheme in context. It also considers the effects of effluent and solids reuse and subsequent effects on soils and surface and groundwaters.

Specifically this section reviews:

- Methods to estimate effluent production.
- Impacts to soils.
- Impacts to surface waters.
- Impacts to groundwaters.

It does not discuss:

- Community amenity issues.
- Biodiversity.
- Flora and fauna.
- Cultural / natural heritage.

This section focuses on the facilities that are subject to licensing, the methods used to determine acceptable effluent and solid by-product reuse rates, the methods used to determine nutrient removal rates by cropping and recommended monitoring requirements.

4.1. Piggeries

4.1.1. ANZEEC Guidelines

Under the National Water Quality Management Strategy “Draft Effluent Management Guidelines for Intensive Piggeries” were developed. These guidelines list a number of criteria for calculating land requirements, including:

- Susceptibility to surface runoff and erosion.
- Potential effect on, and of, groundwater and surface water.
- Climatic conditions.
- The nature of pasture or crop grown.
- Pastoral, agricultural and horticultural practices.
- The properties of soils.
- Trace element loading.

- The quality and quantity of effluent. The maximum potential life of reuse areas is determined by the phosphorus sorption capacity of the soil and predicted salt accumulation.

These guidelines also identify soil properties that are suitable for reuse. When calculating loading rates the hydraulic loading, the nutrient loading or balance (N, P, K) or the salt loading rates will be the most limiting. In general, the maximum nutrient application rates will be 50 to 200 kg N/ha/yr depending on the climate, soil, vegetation, land use and effluent management. So in areas where groundwater has been assigned an environmental value of drinking water or ecosystem protection 1 ha is required for each 6 - 24 pigs.

A nutrient balance can be developed, where losses from the system are:

- The uptake of nutrients by plants harvested.
- Gaseous losses of nitrogen.
- Net accumulation of nutrients in the soil.

These balances allow for seasonal variations in nutrient budgets, including mineralisation and leaching. Long-term nutrient monitoring of the soil and/or soil solution can substitute for the nutrient balance approach.

Effluent application is not recommended for woodlots and grazing systems as they remove very little of the nutrients.

Determination of characteristics of effluent through analysis is encouraged. It is suggested that this include total solids, suspended solids, BOD, COD, organic carbon, electrical conductivity (EC), exchangeable cations, sodium adsorption ratio, pH, total Kjeldahl nitrogen, ammonia nitrogen, phosphorus, potassium, sulphate, metals (zinc and copper), synthetic pyrethroids and pathogens such as salmonella

An irrigation plan should be developed and include:

- Irrigation methods.
- Crop, water and nutrient requirements.
- Application rates.
- Scheduling.
- Design for the collection.
- Storage.
- Utilisation and management of stormwater and tailwater.
- Salt management plan.

Where flood or furrow irrigation is used, terminal ponds should be constructed for the management of tailwaters.

Monitoring is an essential part of an Environmental Management System and/or Plan. The extent of monitoring required should be determined by the piggery and property size, and the environmental sensitivity of the location. Monitoring of effluent quality and volumes for reuse are needed for effective management. An extensive list of guidelines for monitoring and reporting are provided.

4.1.2. National Pollution Inventory (NPI)

The National Pollutant Inventory (NPI) is a publicly accessible database providing information on a geographic basis about specific emissions to the environment. Where an NPI Handbook has been published for an industry, a facility in any Australian state or territory must report its annual emissions for particular substances where it exceeds specified thresholds for these pollutants. An NPI handbook has been prepared for piggeries. The major pollutants likely to be reported are total nitrogen and total phosphorus to water, and ammonia and combustion products to air.

From 1 June 2002, the previously voluntary reporting of pollutant emissions to the NPI became a mandatory requirement in NSW, under the *Protection of Environment Operations (General) Amendment (National Pollutant Inventory) Regulation 2002.*

In NSW, two piggeries reported to the NPI in 2001 – the main pollutant reported was ammonia emissions to air. Pollutant emissions to water are not allowed under licence conditions for piggeries.

4.1.3. The NSW System

The NSW EPA has model licence conditions for developing individual licence conditions for operations. These contain general administrative details, discharges to air and water and applications to land, limit conditions, operating conditions, monitoring and recording conditions, reporting conditions and general conditions. Specific requirements relating to the reuse of effluent and solid by-products include:

- Effluent application must not cause surface runoff.
- The quantity of effluent or solid by-products reused must not exceed the land's capacity for effective utilisation. Effective utilisation includes the use of effluent or solid by-products for pasture or crop production, and the soil's capacity to absorb the nutrient, salt, water and organic material applied.
- If solids are removed from the premises, the licensee must record the date, estimated weight of solid by-products and the identity of the person receiving the solid by-products.

There are no load, concentration and volume or mass limits, provided the above requirements are met.

4.1.4. The Queensland System

Guidelines

The Environmental Code of Practice for Queensland Piggeries (Streeten and McGahan, 2000) is the current code in Queensland.

Approval Process

The approval process for piggeries in Queensland is controlled by the *Integrated Planning Act 1997 (IP Act)*. Essentially, this means that all relevant licences, approvals and permits are issued at the one time under a co-ordinated assessment process. For piggeries, this means land use approval from the local Shire and the issuance of an Environmental Licence under the *Environmental Protection Act 1994 (EP Act)* by the Department of Primary Industries (DPI). While many Shires previously had by-laws covering piggeries, in most cases, the Shires follow the principles stated in DPI (2000) and follow DPI's recommendations in relation to environmental impact.

For all but exceptional circumstances, piggeries are therefore approved on the basis of compliance with the various standard formulae given in DPI (2000). A Groundwater Impact Assessment is usually required and this is a risk assessment of the likelihood of groundwaters being affected. Soil testing on the proposed effluent utilisation area is mandatory. The soil analysis parameters are given in Table 3.

TABLE 3 – STANDARD SOIL TESTS – QLD PIGGERY APPLICATION

Total nitrogen or Total Kjeldahl nitrogen	TN or TKN
Nitrate nitrogen	NO_3^--N
Acid extractable or Colwell extractable phosphorus	Colwell Extractable P
Potassium	K
Exchangeable Sodium Percentage	ESP
Electrical conductivity	$EC_{1:5}$ (1:5 dilution ratio)
pH	pH
Organic carbon	OC

An Environmental Management Plan must also be developed.

4.1.5. The Victorian System

The Victorian Code of Practice Piggeries, Department of Food and Agriculture (1992) is the current code. It specifies minimum standards for new piggeries or where there are substantial modifications to existing piggeries. Requirements cover general (site) requirements, piggery classifications, buffers to sensitive areas, building design requirements, operating requirements and effluent reuse. The *Code of Practice Piggeries* and the companion volume entitled *Guidelines for Siting and Operation of Piggeries* are designed to assist municipal councils, pig producers and planning authorities. Specific effluent reuse issues include:

- The site should have an undulating or flat terrain to minimise soil erosion.
- The soil type should ideally be medium loamy-clay to provide reasonably good drainage and retention of nutrients.
- Piggery effluent is not to be spread on land that is liable to flooding at a frequency exceeding 1 in 5 years.
- Piggery by-products, or stormwater contaminated by piggery by-products, shall not be allowed to leave the property.
- Polluting materials shall not be permitted to enter any groundwater since this may be detrimental to the beneficial use of groundwater or surface water.

- The hydraulic loading of each soil type shall not be exceeded.
- Effluent reuse rates should not exceed the suggested NPK loadings (N – 350 kg/ha/yr, phosphorus – 150 kg/ha/yr, potassium – 200 kg/ha/yr), or such values that are normally accepted as being used by the designated vegetative cover.
- Soil testing shall only be required when the actual reuse are available is less than 1 ha per 35 pigs. If soil testing is required, it determines the permitted hydraulic loading, such that all the effluent applied to the land will be taken up by the soil moisture deficit and/or evaporated through evapotranspiration and that all the nutrients applied will be taken up by the soil deficiency (if any) and the type, or proposed type of vegetative cover.

The 1992 Code is currently under revision and a Draft Code (Hogan, 2000) currently exists. It is understood that this code is undergoing substantial revision prior to release. It is far more detailed than the 1992 Code. Some of the criteria in relation to reuse are:

- To ensure that the application rate of nutrients (particularly nitrogen, phosphorus and potassium) does not exceed the sum of the nutrient uptake rate by plants or the quantity that can be safely stored in the soil.
- To ensure that monitoring is undertaken to provide a means of managing soils and groundwater.
- To ensure the control of run-off and seepage from effluent reuse areas, so as to avoid adverse impacts to surface waters or groundwater via processes such as soil erosion and leaching.

The Draft Code (Hogan,2000) allows for the storage of phosphorus in reuse areas. The document specifies that monitoring is needed to aid in the calculation of correct application rates of fertiliser and to identify adverse environmental impacts requiring corrective action. The draft code indicates that the standard soil tests provided by commercial companies will generally be sufficient. Annual surface soil sampling and biannual soil profile sampling are suggested, along with annual and biannual effluent quality analysis. Monitoring requirements vary substantially from the 1992 Code of Practice where soil testing was only required where the "actual disposal area available" was less than 1 ha for 35 pigs.

It is important to note that these requirements are likely to change as the code is revised

4.1.6. The South Australian System

PIRSA (1998) is the relevant guideline for the development of piggeries in South Australia. The guideline outlines the relevant legislation and the procedures for obtaining an approval. Site selection and effluent treatment systems are described. Environmental objectives are stated. The guideline notes that an effluent spreading plan should be developed for each site. The discussion of reuse of solid and liquid by-products is generic in nature with no specific definitions. Appendix 12 gives two examples for calculation of land needed for effluent reuse. In the sample calculation, the piggery effluent has the following analysis - 2175 mg N/L, 850 mg P/L and 1618 mg K/L. These are very high concentrations. The guideline then suggests that 50% of total N, 70% of total phosphorus and 90% of total potassium will be available to the crop in the year of application. An effluent application rate is calculated using nutrients removed by the crop only matched to the available nutrients applied (i.e. no phosphorus sorption). Consequently, total nutrients applied exceed total nutrients removed. Table 4 gives the South Australian recommendations for soil and groundwater monitoring. No monitoring of soil nitrogen level is included.

TABLE 4 – SOUTH AUSTRALIAN PIGGERY MONITORING PARAMETERS

Monitored Element	Parameters
Soil	Extractable Phosphorus
	Extractable Potassium
	Organic Carbon
	Phosphorus Retention Index
	Salinity
	pH
	Manganese
	Zinc
	Sulfur
Groundwater	Total Salt and Nitrates

4.1.7. The West Australian System

Latto *et al.* (2000) is the relevant guideline for piggeries in Western Australia. This guideline includes legislative requirements and procedures for obtaining a piggery approval. It covers intensive and extensive piggeries including deep-litter systems. For effluent, this guideline refers in several areas to Ryan and Payne (1989) and Kruger *et al.* (1995).

The guideline has a section on treated effluent reuse. Table 5 is used to determine maximum nutrient loading rates. The guideline also recommends that the degradable organic matter loading rate should not exceed 30 kg BOD/ha/day to avoid offensive odours. Heavy metals applications should not exceed Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines for fresh and marine waters. It is recommended that piggeries develop environmental management plans.

TABLE 5 – SUGGESTED MAXIMUM PHOSPHORUS AND NITROGEN PIGGERY APPLICATION RATES (WA)

Vulnerability Category	Soil Description	Max available phosphorus (as P) loading (kg/ha/yr)*	Max available nitrogen (as N) loading (kg/ha/yr)
A	Coarse sandy soils / gravels (PRI**<10) draining to surface waters with moderate / high risk of eutrophication	10	140
B	Coarse sandy soils / gravels (PRI<10) draining to waters with a low risk of eutrophication.	20	180
C	Loams / clay soils (PRI>10) draining to waters with moderate / high risk of eutrophication	50	300
D	Loams / clay soils (PRI>10) draining to waters with a low risk of eutrophication.	120	480

Source: Soil vulnerability and loading rates derived from Water and Rivers Commission's Water Quality Protection Note, November 1998.
* Phosphorus is the rate-limiting nutrient. Maximum manure application assumes no additional nutrient sources (e.g. fertilisers).
** PRI means phosphorus retention index.

4.1.8. The Tasmanian System

The Department of Primary Industries, Water and Environment (DPIWE) provided advice on piggery regulation in Tasmania. Local government regulate piggeries as environmentally relevant activities. State departments have little input to the environmental performance of piggeries.

A document entitled "Environmental Guidelines for Piggeries" (Brennan & Howett, 1990) exists. It provides very basic information on effluent and solid by-products reuse. Soil type; soil porosity; depth to groundwater; rainfall; output and land available need to be considered. Effluent application rates for pasture should not exceed 500 kg/ha/yr for nitrogen (N) and 300 kg/ha/yr for potassium (K). As a guide, fresh undiluted manure from 60 average sized pigs (45 kg liveweight) should be spread over at least one hectare per year.

4.2. Feedlots

A beef feedlot is a confined area with watering and feeding facilities where cattle are completely hand or mechanically-fed for the purpose of production (ARMCANZ, 1997). Feedlot operations include feedstock storage, feeding systems, animal housing, manure and effluent removal/storage and effluent and manure treatment.

4.2.1. National Feedlot Accreditation System (NFAS)

The National Feedlot Accreditation Scheme (NFAS) is an industry self-regulatory quality assurance scheme that was initiated by the Australian Lotfeeders Association (ALFA). It is managed by an industry committee: the Feedlot Industry Accreditation Committee (FLIAC).

The scheme aims to ensure that grain fed beef from accredited feedlots is produced in accordance with approved standards and procedures and will consistently meet the expectations and specifications of customers. Environment protection procedures are an integral component of the scheme. Within NFAS, individual operations need to address the requirements of the National Guidelines and Codes of Practice.

To be accredited, each feedlot operator must:

- Develop site specific documented procedures meeting the requirements of the industry standards
- Maintain records demonstrating adherence to these procedures for all cattle prepared at the feedlot
- Undergo a third party audit of these procedures, the associated records and the feedlot facilities.

4.2.2. National Guidelines and Codes of Practice

Before 1992, there was no nationally recognised guideline or code of practice for feedlots that was endorsed by all stakeholders. Following a series of workshops involving all parties, national guidelines were developed. These guidelines were revised in 1997 (SCARM, 1997).

The intent of these guidelines is to provide a framework of acceptable principles for the establishment and operation of feedlots in Australia. The guidelines specify acceptable standards for good management practice across Australia. They include a definition of a feedlot and the definition of standard cattle units (SCU).

The guidelines provide a number of generic environmental performance objectives aimed at protecting community amenity, land resources, surface waters and groundwaters. They also discuss the various components of a typical feedlot and provide design objectives and concepts. They provide information on effluent and manure utilisation areas and terminal systems. In part, the guidelines state:

Effluent and Manure Utilisation Systems

> ***Objective****: To employ crops/pastures and soils to effectively utilise or sustainably assimilate the nutrients, salts, organic matter and water contained in feedlot effluent and manure.*
>
> ***Design Concept****: The area of land required to enable utilisation of the effluent and/or manure applied under a given crop/pasture regime should be calculated using water, nutrient and salt balances and a critical organic loading rate. Crops/pastures need to be harvested and removed from utilisation areas to prevent nutrient build-up. Where nutrients and salts are not taken up in plant growth and removed, their sustainable assimilation by the soil must be demonstrated.*
>
> ***Design Calculation****: The annual loading rate for each of the constituents of the effluent and manure (e.g. nitrogen, phosphorus, salt and hydraulic load) should be calculated. The minimum area required for effluent utilisation will be the largest calculated for any individual constituent.*

Terminal Systems

> ***Objective****: To collect and recycle all irrigated effluent tailwater and to manage contaminated stormwater runoff from the effluent irrigation area, so as not to pollute waters.*

4.2.3. National Beef Feedlotting Code of Practice

The development of the National Beef Cattle Feedlot Environmental Code of Practice was initiated by ALFA in early 1998 to address the environmental legislative requirements of all States and Territories. ALFA identified the need to draw together divergent opinion on standards of environmental management across Australia into a single approved document that would form the basis of future environmental activity.

The National Beef Cattle Feedlot Environmental Code of Practice provides environmental performance objectives, operational objectives and details of practices that feedlot management and staff can use to achieve compliance with the environmental duty of care. It also provides industry agencies, the community and regulatory authorities with benchmarks against which to assess the industry's performance. FLIAC has adopted this report as a replacement for "Code of Practice - Protection of the Environment" (ALFA, n.d.). All NFAS accredited feedlots must now comply with the National Beef Cattle Feedlot Environmental Code of Practice.

To demonstrate compliance with the National Beef Cattle Feedlot Environmental Code of Practice, each feedlot operator will:

- Document clear and achievable environmental objectives, performance indicators for their operational practices and monitoring programs.
- Ensure that feedlot management is aware of and adhere to their environmental legislative requirements.
- Ensure that all employees are aware of and adhere to their environmental management responsibilities.

- Develop procedures to reduce the potential for environmental harm to occur and provide adequate training for employees.
- Monitor environmental performance on an annual basis, or as required by the appropriate regulatory authority,
- Audit environmental operational practices to identify opportunities for improvement against performance indicators, incorporating any such opportunities in future environmental operational practices.
- Maintain an awareness of current and developing industry wide practices to achieve the objectives of the code.

The National Beef Cattle Feedlot Environmental Code of Practice notes that monitoring, recording and reporting are essential components of the system. It describes a number of environmental management practices. In relation to effluent reuse, the National Beef Cattle Feedlot Environmental Code of Practice states:

Effluent Management

Objective:

- *To store and effectively utilise the water, nutrients, salts and organic matter in effluent captured from the controlled drainage areas of the feedlot and collected tailwaters, in a manner that safeguards animal and human health.*

Operational Practices:

- *Clean and maintain sedimentation systems and holding ponds to maintain the capacity, freeboard and impermeability.*
- *Apply effluent to utilisation areas at rates that maintain acceptable nutrient, salt and water balances.*
- *Apply effluent only during favourable weather conditions.*
- *When applying effluent it must not enter natural watercourses, groundwater or neighbouring properties.*
- *Reuse effluent in other feedlot activities, wherever practical.*

Performance Indicators:

- *The productivity of effluent utilisation areas is maintained or enhanced with no demonstrated build up of nutrients or salts above soil holding capacity.*
- *The quality and integrity of adjacent groundwater and surface water are maintained.*
- *Justifiable odour emanating from effluent storage and utilisation areas be kept to a minimum.*
- *No irrigation tailwaters or spray drift onto neighbouring properties during effluent application.*
- *No prolonged ponding of applied effluent after application.*

Monitoring and Recording:

- *Record dates, areas of effluent application and application rates.*
- *Sample and test effluent utilisation areas to demonstrate soil attributes are maintained or enhanced.*
- *Monitor and record the quality and integrity of adjacent groundwater and surface waters, as required.*
- *Record date, time and duration of any effluent overflow outside the controlled drainage area, including the by-washing of the effluent pond.*

- *Collect and analyse samples of effluent discharge for BOD_5, total P, TKN, EC and pH.*
- *Record complaints about odour emanating from effluent storage and utilisation areas, their cause and action taken.*
- *Record incidents of and complaints about effluent entering natural watercourses or neighbouring properties, their cause and actions taken.*

The National Beef Cattle Feedlot Environmental Code of Practice is used as a reference document for the NFAS accreditation system (see Section 4.2.1).

4.2.4. National Pollutant Inventory (NPI)

An NPI handbook has been prepared for cattle feedlots. The major pollutants likely to be reported are total nitrogen and total phosphorus to water, and ammonia and combustion products to air.

In NSW, six feedlots reported to the NPI in 2001 – the main pollutant reported was ammonia emissions to air. Pollutant emissions to water are not allowed under licence conditions for feedlots.

From 1 June 2002, the previously voluntary reporting of pollutant emissions to the NPI became a mandatory requirement in NSW, under the *Protection of Environment Operations (General) Amendment (National Pollutant Inventory) Regulation 2002.*

4.2.5. The NSW System

Guidelines

Following the development of the national feedlot guidelines, NSW Agriculture prepared the Feedlot Manual (NSW Agriculture 1997). This comprehensive document covers all aspects of feedlot establishment and operation. Recently, a document covering smaller feedlots only was issued (NSW Agriculture, 2001). This document requires:

- Terminal pond systems for effluent reuse areas.
- That effluent irrigation be load limited for nitrogen, phosphorus, BOD, salts and hydraulic loading. The design criteria for hydraulic loading should be determined by the rainfall in a 90 percentile wet year (the wettest year in 10 based on at least 40 years of historical or simulated data for the local area). Effluent irrigations should match the requirements of the crops and pastures grown.
- Soil testing requirements are listed on page 40 of the manual.
- Manure application rates are specified for various cropping systems. If these rates are exceeded, soil testing is required. Page 43 of the manual.

Model licence conditions for the reuse of effluent and solid by-products for cattle feedlots are the same as for piggeries (refer to Section 4.1.3).

4.2.6. The Queensland System

Guidelines

In 2000, the Queensland Cattle Feedlot Advisory Committee published a Reference Manual for the Establishment and Operation of Beef Cattle Feedlots (Skerman, 2000). The Reference Manual includes sections on:

- Definitions.
- Approval Process.
- Site Selection.
- Separation Distances.
- Feedlot Design.
- Feedlot Hydrology (drains, sedimentation basins, holding ponds).
- Manure Handling and Composting.
- Effluent and Manure Utilisation.
- Carcass Disposal.
- Site Rehabilitation.

The section of the manual covering effluent and manure utilisation provides detailed background and tools to design effluent and manure utilisation systems. In most cases, the approval process becomes the simple application of a few standard formulae.

Appendix D of the Reference Manual provides guidance on sampling techniques and recommends analysis parameters for soils, water, effluent, sludge and manure. Table 6 gives the recommended parameters.

TABLE 6 – QUEENSLAND FEEDLOT MONITORING PARAMETERS

Monitored Element	Parameters
Soil	pH Electrical Conductivity $_{1:5}$ Total Nitrogen or Total Kjeldahl Nitrogen and Nitrate-Nitrogen Total Phosphorus or Colwell Phosphorus Exchangeable Sodium Percentage Organic Carbon Chloride
Water and effluent	pH Electrical Conductivity Total Phosphorus Ortho-Phosphate Sodium Adsoprtion Ratio Total Nitrogen or Total Kjeldahl Nitrogen, Ammonium-Nitrogen and Nitrate-Nitrogen Potassium
Sludge and Manure	pH Electrical Conductivity Total Nitrogen or Total Kjeldahl Nitrogen, Ammonium-Nitrogen and Nitrate-Nitrogen Total Phosphorus Total Sodium Total Calcium Total Magnesium Total Carbon Potassium

Approval Process

The approval process for Queensland feedlots is controlled by the *Integrated Planning Act 1997 (IP Act)*. All relevant licences, approvals and permits are issued at the one time under a co-ordinated assessment process. This includes land use approval from the local Shire and the issuance of an Environmental Licence under the *Environmental Protection Act 1994 (EP Act)* by DPI. In most cases, the Shires follow the principles stated in the Reference Manual and DPI's recommendations in relation to environmental impact, although some shires apply their own by-laws.

For all but exceptional circumstances, feedlots are approved through compliance with the various standard formulae given in the Reference Manual.

A risk assessment-based Groundwater Impact Assessment is usually required. Soil testing on the proposed effluent utilisation area is mandatory. The parameters for analysis are given in Table 7.

TABLE 7 – STANDARD SOIL TESTS – QLD FEEDLOT APPLICATION

Nitrate + Nitrite N	NO_3^--N + NO_2^-–N
Total Phosphorus	TP
Colwell Extractable Phosphorus	Colwell Extractable P
Potassium	K
Exchangeable Sodium Percentage	ESP
Electrical Conductivity	$EC_{1:5}$ (1:5 dilution ratio)
pH	pH
Organic Carbon	OC

4.2.7. The Victorian System

The Victorian Code for Cattle Feedlots (1995) is the applicable guideline for cattle feedlots in Victoria. This document outlines the approval process and relevant legislation. It provides guidance on siting, design and management of cattle feedlots. Formulae are provided to calculate adequate separation distances and holding pond volumes.

The code requires lot feeders to develop a Waste Management Plan (WMP). This provides the basis for the management and use of all of the solid and liquid by-products of the feedlot. The WMP must demonstrate that the feedlot by-products will be applied to land or otherwise used in a manner that is environmentally sustainable having regard to existing and proposed nutrient levels, salinity and hydrological considerations. The code states that "the complexities of waste management preclude definitive and detailed statements to cover all situations, and this aspect of the plan should be developed in consultation with the code."

The code provides some guidance on monitoring parameters for soils, effluent and groundwater. They are given in Table 8.

TABLE 8 – VICTORIAN FEEDLOT MONITORING PARAMETERS

Monitored Element	Parameters
Soil	Olsen P (mg/kg)
	Skene K (mg/kg)
	Nitrates and Total N (mg/kg)
	Exchangeable Na
	EC
	pH
	CEC
	Total Exchange Basis
	Dispersion (Emerson test)
Groundwater	pH
	EC
	Olsen P (mg/L)
	Nitrates
Surface Waters	Olsen P (mg/L)
	Nitrates (mg/L)
	EC
	BOD

4.2.8. The South Australian System

PIRSA (1994) is the relevant guideline for the development of feedlots in South Australia. This document provides only generic guidance on the reuse of solid and liquid effluent with no specific definitions, methods for calculating loading rates, or sustainability criteria.

4.2.9. The West Australian System

In Western Australia, DEP (1993) was the applicable guideline. Although still in draft form, DEP (2000) has recently replaced DEP (1993). The guidelines explain the method for obtaining approval in Western Australia and the legislation that applies. General design and operation principles are outlined.

DEP (2000) specifies maximum nutrient loading rates. Specifically, the guidelines state:

> *The application of nutrient-rich by-products should not exceed the total nutrient load outlined in Table 9. The nutrient loading to land is a cumulative loading from all sources, i.e. solid manures, liquids and any artificial fertiliser added.*

4.2.10. The Tasmanian System

The Department of Primary Industries, Water and Environment (DPIWE) provided advice on the Tasmanian situation. There are no state-specific feedlot guidelines in Tasmania. This is possibly because there is only one feedlot in Tasmania of any significance. It is regulated as a level 2 activity by the environment division of DPIWE. This feedlot is also accredited under the NFAS scheme.

TABLE 9 – MAXIMUM PHOSPHORUS AND NITROGEN FEEDLOT APPLICATION RATE CRITERIA TO PROTECT WATER QUALITY (WA)

Vulnerability Category	Soil Description	Max available phosphorus (as P) loading (kg/ha/yr)*	Max available nitrogen (as N) loading (kg/ha/yr)	Maximum manure application rate (t/ha/yr)*
A	Coarse sandy soils / gravels (PRI**<10) draining to surface waters with moderate / high risk of eutrophication	10	140	1.2
B	Coarse sandy soils / gravels (PRI<10) draining to waters with a low risk of eutrophication.	20	180	2.6
C	Loams / clay soils (PRI>10) draining to waters with moderate / high risk of eutrophication	50	300	6.1
D	Loams / clay soils (PRI>10) draining to waters with a low risk of eutrophication.	120	480	14.7

Source: Soil vulnerability and loading rates derived from Water and Rivers Commission's Water Quality Protection Note, November 1998.

- * Phosphorus is the rate-limiting nutrient. Maximum manure application assumes no additional nutrient sources (e.g. fertilisers).
- ** PRI means phosphorus retention index.

4.3. Generic Guidelines

4.3.1. NSW EPA Use of Effluent in Irrigation Guidelines (Draft)

The NSW EPA Draft Use of Effluent in Irrigation Guidelines cover best practices and procedures for establishing an effluent irrigation system. Effluent from intensive livestock facilities is covered by the document.

It is intended that the environmental best management practices outlined in the effluent irrigation guideline will provide a benchmark for all regulatory authorities administering effluent reuse by irrigation schemes. It is noted that the document is an environmental guideline, not a design and operations manual. It is also noted that to achieve sustainability, "a program of continuous monitoring and progressive modification might be necessary".

The guidelines are organised as follows:

- Section 1 outlines the broad scope, objectives and procedures for establishing an effluent irrigation system,
- Section 2 provides guidance on the site planning for an effluent irrigation system. It includes general site requirements and particular requirements for soils, surface

water, groundwater and flooding. This section could be used as a reference source for producers seeking guidance on site assessment.

- Section 3 describes important characteristics of effluent to consider in establishing effluent irrigation systems. This summarises effluent characteristics that should be used to assess potential environmental effects arising from effluent irrigation.
- Section 4 outlines irrigation system design considerations. This section summarises issues to consider when assessing the sustainability of an irrigation area, including water and nutrient balance, crop removal rates, salt balance and heavy metals.
- Section 5 outlines irrigation system operation considerations. It summarises issues to consider in maintaining the sustainability of an irrigation area, including site management plans, control and monitoring systems, monitoring requirements and safety considerations.
- Section 6 summarises statutory requirements for an effluent irrigation system.

The following principles are suggested by this document to determine whether an effluent reuse system is sustainable:

Resource Use: Potential resources in effluent, such as water, plant nutrients and organic matter, should be identified, and agronomic systems developed and implemented for their effective use.

Protection of Lands: An effluent irrigation system should be ecologically sustainable. In particular, it should maintain or improve the capacity of the land to grow plants, and should result in no deterioration of land quality through soil structure degradation, salinisation, water logging, chemical contamination or soil erosion.

Protection of Groundwater: Effluent irrigation areas and systems should be located, designed, constructed and operated so that the current or future beneficial uses of groundwater do not diminish as a result of contamination by the effluent or run off from the irrigation scheme or changing water tables.

Protection of Surface Waters: Effluent irrigation systems should be located, designed, constructed and operated so that the surface waters do not become contaminated by any flow from irrigation areas, including effluent, rainfall run off, contaminated sub surface run off, or contaminated groundwater**.**

Prevention of Public Health Risk: The effluent irrigation scheme should be sited, designed, constructed and operated so as not to compromise public health. In this regard, special consideration should be given to the provision of barriers that prevent human exposure to pathogens and contaminants.

Community Amenity: The effluent irrigation system should be located, designed, constructed and operated to avoid unreasonable interference with any commercial activity or the comfortable enjoyment of life and property off-site, and where possible to add the amenity. In this regard, special consideration should be given to odour, dust, insects and noise.

4.3.2. Victorian Manure Guidelines (Draft)

The Draft Victorian Manure Guidelines were developed to provide guidance on the beneficial use of animal manure solids. Although the guidelines are still under development, they include sections on site assessment, maximum contaminant concentrations, nutrient and contaminant loadings and nutrient removal. The approach taken in these guidelines is very similar to that presented in other documents so this guideline will not be referenced further.

4.3.3. Composting Standards

The first measure of the quality of compost is compliance with the relevant Australian Standard or equivalent. Three Australian Standards relate to products containing composts:

- AS4454 – Compost, Soil Conditioners and Mulches.
- AS3743 – Potting Mixes.
- AS 4419 – Soils for Landscaping and Garden Use.

The Standards are a tool to facilitate the sustainable recycling of organic materials by guaranteeing that products are consistent in quality and safe to use; and by showing customers that compost producers are committed to quality control.

4.4. Implementation of Research in Regulations

There is clearly a lack of a defined path for upgrading Codes of Practice and Guidelines, with many of these documents being very outdated and/or very conservative because of a lack of knowledge and application of the "Precautionary Principle". However many of the more recently produced codes for intensive animal industries are planning 5-year reviews and upgrades of the publications. It would be beneficial to include relevant, peer reviewed findings from current and future research in regular upgrades of codes and guidelines. National Code or Guidelines could provide a vehicle for this process. The feedlot industry currently has a National Code and the pig industry is developing National Guidelines. It is important however, that these documents are regularly updated.

5. SUSTAINABLE EFFLUENT AND MANURE REUSE

Figure 1 provides a framework to achieve sustainable effluent and by-product reuse for piggeries and cattle feedlots. It assumes that environmental monitoring requirements for piggery or feedlot reuse areas should be tailored to match the potential risk to the environment. Consequently, the framework provided uses a **risk assessment process** to decide monitoring requirements. The level of risk is a function of the environmental vulnerability of the site, the quantity of water, nutrients and salt for reuse and the design and management of the reuse areas.

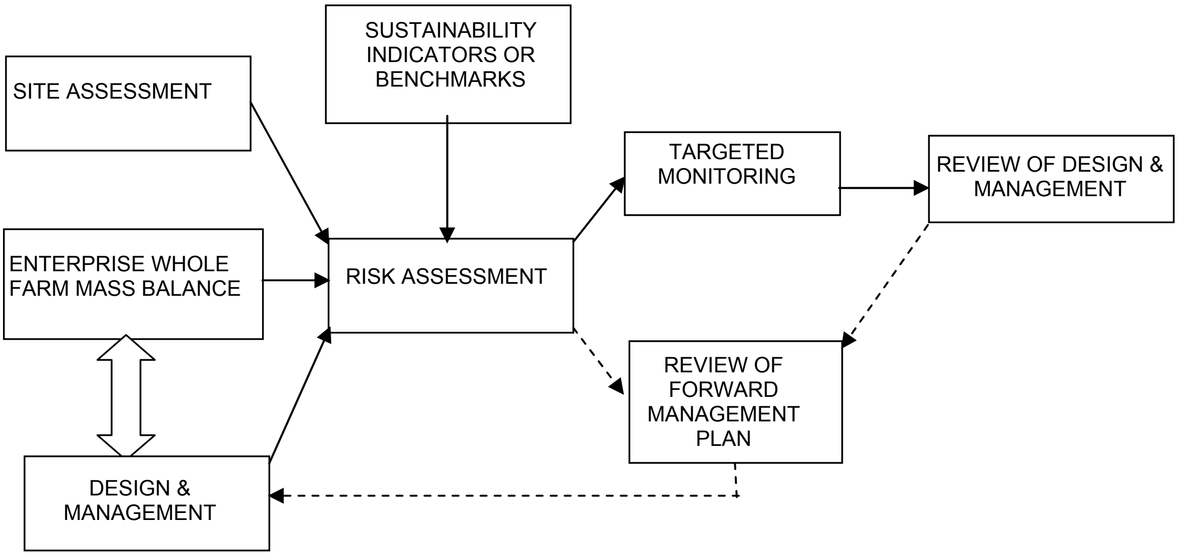


FIGURE 1 – MANAGING SUSTAINABLE REUSE FOR PIGGERIES AND CATTLE FEEDLOTS

The first stage in an environmental risk assessment is a site assessment. This identifies any resources that are vulnerable to any adverse impacts from reuse, as well as the site factors that could influence reuse. It includes an evaluation of the soils of the site, the nearby surface water resources, groundwater depth and quality, the climate of the area (rainfall, evaporation etc), the land area available for reuse and the type and expected yield of the crops or pastures that could be grown. Different sections of one reuse area can have different vulnerabilities depending on the natural resources of these areas. Also, if there are multiple reuse areas on a property these may have different vulnerabilities. Thus, the process identified in Figure 1 should be applied separately to each individual reuse area. Further details of the site assessment stage of the risk assessment are provided in Section 10.

Next, the quantity of nutrients and salts for reuse, and the design and management of reuse areas must be examined. **An enterprise whole farm mass balance can be used to estimate the mass of nutrients for reuse and the mass removed from reuse areas, stored in the soil or lost to the environment.** The mass of nutrients and salts produced by an enterprise can be taken from standard 'text book' values, estimated using mass balance principles or calculated from the measured salt and nutrient concentrations in the by-products and the quantity of effluent or solids applied (for operating enterprises). The mass of nutrients and salt removed can be estimated from 'text book' or analysed nutrient composition data multiplied by measured or estimated plant yields. Further details on nutrient and salt estimation methods are provided in Section 6.

The standard of design and management of the reuse areas influences the risk of environmental harm. Where environmentally vulnerable resources are identified or where water, nutrient or salt loading rates are high, good design and management can significantly decrease the risk to the environment. However, poor design or management can cause environmental harm even when resources are not particularly vulnerable. Further details on the design and management aspects of reuse areas are provided in Section 7.

The actual risk assessment decides if adverse environmental impacts are likely, considering the combined effect of the environmental vulnerabilities of the site, the quantity of water, nutrients and salt for reuse and the design and management of the reuse areas. The sustainability indicators included in Section 8 provide the evaluation criteria for the risk assessment (which follows as Section 11).

Theoretically, the simplest measure of sustainability is a match between nutrient application rate to a land area and the nutrient removal by plant harvest from that area. This will never occur in reality due to a number of other factors, including:

- The soil/by-product dynamics after application.
- Nutrient availability of by-products.
- Losses such as nitrogen volatilisation.
- Storage of nutrients such as phosphorus in the soil.
- Leaching of salts through the soil profile.
- Exports of some elements in surface water runoff.
- The need to address pre-existing soil nutrient deficiencies (to bring these up to normal agronomic levels).

Thus, application rates need to be closely matched to estimated uptake rates, plus *acceptable* storage and losses of nutrients for a system to be sustainable.

Sustainability indicators measure the effects on the environment of nutrients applied to reuse areas. These indicators are not absolute measures, so a process must be established to assess what the indicators tell about the site. **A risk assessment is the most appropriate process to interpret the effects of reuse on the environment.** This provides flexibility to assess a broad range of sites without compromising the accuracy of the assessment. A suggested matrix process is provided in Section 11. It decides the environmental risk profile and consequently the scope of environmental monitoring required. This leads naturally into Section 12, which outlines suggested monitoring parameters and frequencies.

To determine the environmental risk and consequently the scope of environmental monitoring required, a matrix process is suggested. This leads naturally into suggested monitoring parameters and frequencies. The following sections of the document provide guidance for a piggery or cattle feedlot the operator to complete a matrix and determine their level of environmental risk.

In Section 6 and 7 the risks associated with the design and management of the reuse area are determined. These include a knowledge of the nutrients in effluent and manure available for reuse (Section 6), knowledge of the size of land area available and application rate of nutrients (Section 7) and the risk associated with the application method of effluent and solids (Section 7).

In Section 10 a site vulnerability assessment needs to be conducted. This is assessment of the reuse against the soils of the site (texture, depth, slope, soil dispersion, nitrogen levels and phosphorus levels), surface water (water quality and flood potential) and groundwater (depth to groundwater and soil type).

A matrix is developed by multiplying the risks associated with the design and management of the reuse area, against the site vulnerability assessment (soils, surface water and groundwater).

Where significant environmental risk exists, a piggery or feedlot operator could improve the design or management of the reuse practices or the reuse area to reduce the likelihood of adverse impacts through a Review of the Forward Management Plan (see Section 13). These changes would necessitate a reassessment of risk and monitoring requirements.

The targeted monitoring step may also identify the need for a Review of the Forward Management Plan. Targeted monitoring measures compliance with sustainability indicators where significant environmental risk is identified. The results need careful interpretation and reporting to EPA. EPA evaluates the level of environmental performance and if necessary negotiates a Pollution Reduction Program (PRP) or Load Reduction Agreement (LRA) if in LBL with the licensee to lift environmental performance to an agreed level and within an agreed time frame. An LRA is a very good option since the fees that would otherwise be paid via LBL are spent on improvements to achieve better environmental performance. EPA is keen to see gains made where it is economically feasible to do so, and where there is an environmental imperative.

6. EFFLUENT AND MANURE PRODUCTION (MASS BALANCE)

Nutrients and salt management for effluent and manure reuse areas requires quantification of these constituents. Under the LBL protocol, the mass of nutrients and salts for reuse can be determined in a number of ways, including measuring the volume and concentration of nutrients in effluent, using emission data or using mass balance calculations.

This section details methods for estimating or measuring the nutrient content of manure from piggeries and cattle feedlots. It examines a range of different quantification methods for both piggeries and cattle feedlots. While this section provides general guidance on the amount of nutrients produced by piggeries and cattle feedlots, it does not attempt to distinguish between the amount partitioned to the effluent and solids components. This will depend upon the climate, design and management features of individual operations. The fate of nutrients, including their partitioning to effluent and solids can be estimated using predictive models (such as PigBal, BeefBal and MEDLI). These models are discussed in detail in this section.

The end of this section provides a risk assessment process for evaluating the environmental impacts of the knowledge of the amounts of nutrients available for reuse.

6.1. Estimating Nutrients and Salts in Piggery Effluent & Solids – Concentration and Quantity Method

Table 10 and Table 11 show typical data for the composition of both piggery effluent and solids. These results show a wide variation in the reported data. This is due to the wide variation of design, management, diets and climatic conditions for different piggeries. Thus “typical or average” pond supernatant (irrigation water) and pond sludge concentrations of nutrients and salts cannot be provided and should not be used. Thus, it is advisable to analyse representative samples of effluent and solids for an operation to provide an indication of the quality of the product in terms of nutrients. These measured values are unlikely to change substantially from year-to-year, provided design and management of the operation does not change.

All data in Table 10 is measured from the final pond, which would most often be used for drawing effluent for irrigation.

For an operating piggery, the concentration of elements in the effluent or solid by-products and the total quantity of effluent or solid by-products for reuse provides a measure of the mass of any element available for reuse. This is usually the best method to determine application rates, providing representative samples of effluent or solid by-products are collected and the mass or volume applied is accurately known.

Other methods to determine application rates include the use of “standard values” or mass balance principles. These are detailed in the following Section 6.3.

Methods for quantifying effluent streams and manure mass are provided in Sections 7.1.2 and 7.1.4.

TABLE 10 – CHARACTERISTICS OF PIGGERY POND EFFLUENT, AVERAGE AND (RANGE)

Element	Units	Effluent at Work[1]	DPI Qld – 1994[2]	DPI Qld – 1999[3]
Dry Matter	mg/L	3623	4458 (1240 – 12600)	7900 (1100 – 44300)
Volatile Solids	mg/L	1809	1809 (220 – 4400)	1640 (480 – 5290)
Ash				
pH		8.0		8.0 (7.0 – 8.7)
COD	mg/L			
Total Nitrogen or {TKN}	mg/L	{384}	{654 (158 – 1731)}	584 (158 – 955)
Ammonium Nitrogen	mg/L	249	490 (105 – 1288)	144 (25- 243)
Total Phosphorus	mg/L	44	55.9 (11.0 – 132.0)	69.7 (19.3 – 175.1)
Ortho-Phosphorus	mg/L	28.5	27.0 (3.0 – 91.0)	16.3 (2.4 – 77.9)
Potassium	mg/L		616 (97 – 1845)	491 (128 – 784)
Sulphur			22 (9 – 50)	
SO_4	mg/L			47.6 (13.3 – 87.2)
Copper	mg/L			0.09 (0.00 – 0.28)
Iron	mg/L			0.56 (0.09 – 1.61)
Manganese	mg/L			0.02 (0.00 – 0.05)
Zinc	mg/L			0.47 (0.16 – 1.27)
Calcium	mg/L		18 (7 – 31)	20.6 (7.3 – 41.2)
Magnesium	mg/L		33 (8 – 108)	25.0 (6.6 – 72.3)
Sodium	mg/L	603	603 (103 – 2870)	399 (41 – 1132)
Chloride	mg/L	810	810 (269 – 1950)	19.1 (3.6 – 34.4)
Conductivity	dS/m		7.5 (2.2 – 14.7)	6.4 (2.5 – 11.7)

1. Effluent at Work – Samples from piggeries in New South Wales, Queensland and Western Australia.
2. DPI Qld (1994) – Samples from 10 piggeries in southern Queensland.
3. DPI Qld (1999) – Samples from 10 piggeries in southern Queensland

TABLE 11 – CHARACTERISTICS OF IN-SITU PIGGERY POND SLUDGE, AVERAGE AND (RANGE)

		Effluent at Work (mg/l)[1]	DPI Qld – 1994 (mg/kg)[2]	DPI Qld – 1999 (mg/kg)[3]
Dry matter	% w.b.		17 (6 – 34)	13 (6.9 – 17.1)
Volatile solids	% w.b			6.9 (5.3 – 9.5)
Ash	% d.b			
pH		7.3	7.4 (7.1 – 8.0)	
Carbon	%		12350 (850 – 20160)	28.4 (22.5 – 37.1)
Total Nitrogen or (TKN)	mg/kg	{2617}	{4510 (1750 – 7120)}	3430 (2840 – 4020)
Ammonium Nitrogen	mg/kg	1156		2532 (1472 – 4422)
Total Phosphorus	mg/kg	1696	4720 (560 – 8640)	4710 (2830 – 5900)
Ortho-Phosphorus	mg/l	1082		
Potassium	mg/kg		650 (130 – 2780)	750 (270 – 1330)
Sulphur	mg/kg			1990 (1530 – 3080)
Copper	mg/kg	25	81 (0 – 247)	1062 (343 – 1823)
Iron	mg/kg			1120 (520 – 2210)
Manganese	mg/kg			1035 (786 – 1389)
Zinc	mg/kg			3184 (2184 – 3698)
Calcium	mg/kg	2210		7120 (4280 – 10400)
Magnesium	mg/kg			1920 (1000 – 3190)
Sodium	mg/kg	108		530 (150 – 1400)
Selenium	mg/kg			0.47 (0.07 – 2.41)
Chloride	mg/kg	232	500 (180 – 1770)	
Conductivity	dS/m	8.5	11.4 (6.3 – 16.5)	
SAR				

1. Effluent at Work – Samples from piggeries in New South Wales, Queensland and Western Australia.
2. DPI Qld (1994) – Samples from 10 piggeries in southern Queensland.
3. DPI Qld (1999) – Samples from 10 piggeries in southern Queensland

6.2. Estimating Nutrients and Salts in Piggery Effluent & Solids – Emissions Data

Another method for estimating nutrients and salts in effluent and solids is through the use of emission factors. An emission factor is an estimated pollutant emission rate relative to the level of readily measurable activity. Such factors can be used under the LBL Load Calculation Protocol and the National Pollution Inventory.

Manure estimation research by the Department of Primary Industries Queensland formed the basis for the Standard Pig Units (SPU) concept. The Environmental Code of Practice for Queensland Piggeries (Streeten and McGahan, 2000) defines an SPU as: *The unit of measurement for determining the size of a pig production unit in terms of its waste output. One SPU produces an amount of volatile solids equivalent to that produced an average size grower pig (approximately 40 kg)*. Although the SPU multiplier for each class of pig is based on volatile solids production, it provides similar multipliers between pig classes for other elements (total solids, nitrogen, phosphorus and potassium). Thus, with typical pig diets used in Australia, one SPU excretes about 108 kg of total solids, 90 kg of volatile solids, 18 kg of ash, 9.2 kg of nitrogen, 3.0 kg of phosphorus and 2.4 kg of potassium annually.

One SPU excretes about 108 kg of total solids, 90 kg of volatile solids, 18 kg of ash, 9.2 kg of nitrogen, 3.0 kg of phosphorus and 2.4 kg of potassium annually.

Further mass balance principles are required to estimate the amount of nutrients available for reuse. This will depend on the type of housing system, effluent handling and treatment and climate.

6.3. Estimating Nutrients and Salts in Piggery Effluent & Solids – Mass Balance

This section provides details to estimate the quantity of nutrients and salts in piggery effluent and solids using mass balance principles. Information on predictive models that are based on mass balance principles is also included.

6.3.1. Mass Balance Principles for Piggeries

A mass balance estimates the quantity of nutrients and salts in by-products through the difference between inputs (generally pigs, feed and water) and outputs excluding effluent and solid by-products (pigs, nitrogen volatilisation in sheds). It also provides details of nitrogen losses via ammonia volatilisation and nutrient partitioning in effluent treatment ponds between supernatant and sludge. Each of these elements is important in accurately estimating the quantity of nutrients in the effluent for reuse. Consequently, mass balance models incorporating these principles in their estimations of the quantity of nutrients in the effluent for reuse are discussed after these elements.

Nutrients and salts excreted by pigs can be estimated using predictive models, such as PigBal and MEDLI (see Sections 6.3.2 and 6.3.3) that are based on diet digestibility and mass balance principles. Mass balance principles provide the most accurate method for estimating the amount of nutrients produced by intensive piggeries. They consider different diets, feed use, feed wastage, water quality and use and other factors affecting the amount of manure excreted by each class of pigs, the concentration of constituents and hence the mass of each constituent excreted.

6.3.2. PigBal Model

PigBal 1.0 is an Excel® spreadsheet developed by the Department of Primary Industries, Queensland. It estimates the characteristics of effluent from intensive piggeries. It calculates the total solids (TS), fixed solids (FS), volatile solids (VS), Nitrogen (N), Phosphorus (P), Potassium (K) and salt in the manure from a piggery in which pigs are fed a diet of known composition. PigBal 1.0 uses the Digestibility Approximation of Manure Production (DAMP) (Barth, 1985) to predict the TS, FS and VS and a mass balance approach to estimate the N, P, K and salt in the manure. PigBal has since been upgraded (Version 2.0) to include a more accurate dry matter digestibility approach to predict the solids excreted. The amount of ash excreted is calculated using mass balance principles (mass of ash offered – mass of ash taken up in liveweight gain = mass of ash excreted) in the latter version of the model.

PigBal can assist in the design of effluent treatment facilities and in assessing the environmental sustainability of effluent reuse. It predicts the piggery effluent volume. It also estimates the effluent treatment pond volatile solids loading rate. The mass of nutrients available for reuse in the effluent, sludge and any screenings is calculated from inputs of partitioning and losses (volatilisation of ammonia).

PigBal can cater for all Australian intensive piggery enterprises. It is a useful tool for comparing the effluent production from a traditional farrow-to-finish piggery versus a piggery having an alternate herd structure, for example, a breeder-only or grower unit.

The model has not yet been developed to a fully commercial standard. It is currently used primarily for internal DPI research and assessment. However, a number of consultants now use the model (or its derivatives) to assist in preparing applications for new and expanding piggeries and for developing effluent and manure management strategies for existing piggeries. Copies of the model are available upon request from the Department of Primary Industries, Queensland, on the understanding that the model has not been finalised as not all outputs have been thoroughly validated against measured data from operating piggeries. Nevertheless, for piggeries with very large reuse areas relative to the amount of nutrients for reuse, a desk-top mass balance prepared using PigBal may provide an adequate estimate of nutrients for reuse.

McGahan *et al.* (2001) validated PigBal's predictions of solids and nutrients exiting a conventional flush piggery shed with a series of experiments performed at a 2500 sow farrow-to-finish operation in southern NSW. The experiments involved a section of a finisher shed containing 500 pigs. These shed scale experiments involved the measurement and analysis of the fresh drinking and flushing water, feed usage and effluent exiting the shed. The mass of pigs at the start and end of the trials, the mass of pigs entering or exiting during the trial periods and the mortalities removed were also measured.

Table 12 shows the predicted versus measured results for one of the experiments, which was conducted in September 1999.

TABLE 12 – PREDICTED AND MEASURED OUTPUT RESULTS FOR PIGBAL VALIDATION

	Component								
	TS	**VS**	**Ash**	**N**	**P**	**K**	**Na**	**Ca**	**Mg**
Predict. output (kg)	4,164.7	3,353.9	810.8	411.7	99.5	133.5	16.3	140.2	43.2
Meas. output (kg)	4,164.0	3,355.9	808.1	365.1	97.8	138.3	18.4	120.1	45.3
% Difference (Pred. – Meas.)	-	-0.1	0.3	11.3	1.7	-3.6	-13.2	14.3	4.9

The measured value of nitrogen is approximately 11% lower than the predicted result. Nitrogen loss through ammonia volatilisation both in the shed and during collection could not be measured within that research project. If it is assumed that approximately 10% of the total nitrogen is lost by volatilisation in the shed, then the predictive mass balance method of determining nitrogen excretion via mass balance is accurate. The predicted and measured outputs of phosphorus were very similar.

The predicted TS output could be matched to the measured TS output by setting the feed wastage to 9.0%. Thus the level of feed wastage is still only an approximation. It is however typical of estimated feed wastage values for pigs of this age fed with wet/dry, multi-space feeders as reported by Willis (1999) using the AUSPIG model (Black *et al.*, 1986). Willis (1999) calculated that feed wastage for growing pigs fed pelleted feed in dry multi-space feeders to be 10% and single space wet/dry feeders to be 8%.

The DMDAMP theory and mass balance provides more accurate estimates than using 'historical' textbook values for solids production from pigs. The DMDAMP method considers the amount of feed fed to the animal and its corresponding digestibility to predict effluent production. Textbook values that estimate manure production from the live-weight of the animal and do not take into account feed intake and feed wastage, are likely to over-estimate TS and VS output.

McGahan *et al.* (2001) used the validated DMDAMP theory and mass balance in the PigBal model to predict the solids and nutrient production for different classes of pigs (gilts, boars, gestating sows, lactating sows, suckers, weaners, growers and finishers). Table 13 provides results.

TABLE 13 – PREDICTED MANURE COMPONENT OUTPUT FOR EACH CLASS OF PIG USING DMDAMP AND MASS BALANCE.

Pig Class	TS (kg/yr)	VS (kg/yr)	Ash (kg/yr)	N (kg/yr)	P (kg/yr)	K (kg/yr)
Gilts	197	162	35	12.0	4.6	4.0
Boars	186	151	35	15.0	5.3	3.8
Gestating Sows	186	151	35	13.9	5.2	3.7
Lactating Sows	310	215	95	27.1	8.8	9.8
Suckers	11.2	11.0	0.2	2.3	0.4	0.1
Sow + Litter	422					
Weaner pigs	54	47	7	3.9	1.1	1.1
Grower pigs	108	90	18	9.2	3.0	2.4
Finisher pigs	181	149	32	15.8	5.1	4.1

6.3.3. MEDLI Model

MEDLI® is a Windows® based computer model for designing and analysing effluent treatment and reuse systems using land irrigation. It was developed jointly by the CRC for Waste Management and Pollution Control, the Queensland Department of Primary Industries and the Queensland Department of Natural Resources. MEDLI is a complex, daily-time-step, hydrological simulation model developed to estimate the effluent stream in an enterprise through to the reuse area and predicts the fate of the water, nitrogen, phosphorus, and soluble salts over extended periods. MEDLI is very flexible and can handle a wide range of industries such as piggeries, feedlots, abattoirs, sewage treatment plants, and dairy sheds, and any user-defined effluent stream such as a food-processing factory.

There is no other equivalent model that can follow the effluent stream of a piggery from its estimation through pre-treatment, treatment and reuse. MEDLI is particularly useful as a water balance model to optimise sizing of effluent treatment ponds and reuse areas. MEDLI has been used to calculate sustainable reuse areas of at least 100 piggery developments in Australia.

MEDLI does not model the fate and reuse of nutrients and salts in the solid by-products, such as piggery sludge and feedlot manure.

MEDLI uses the same principles as PigBal of mass balance and diet digestibility to predict the wastestream from the sheds. This methodology has been proven as previously discussed. MEDLI predictions have also been compared against measured pond nutrient composition (N and P) and salinity, providing very good predictions of these components.

MEDLI modelling for piggeries identifies that nutrients and not hydraulic loading limit sustainable effluent reuse rates. With nearly all simulations for piggeries, the depth of effluent is less than 100 mm annually.

Dougherty (1996) reviewed available models to assist in determining future directions for NSW Agriculture in reviewing development applications. A number of models from each category (water, salt and nutrient) were found to be of some value. He concluded that MEDLI provided a comprehensive assessment of all the factors being considered. Although MEDLI has a number of deficiencies (no solids component), it is by far the most comprehensive model available and has been designed specifically for modelling the fate of nutrients and salts in agricultural industries.

6.3.4. Other Predictive Models

MESPRO

Developed in Holland, MESPRO (Aarnink et al., 1992) is a mathematical model for estimating the dry matter content and the amount of nitrogen (total and ammonium), phosphorus, calcium, magnesium and potassium in effluent slurry from fattening pigs. Input variables are animal weight, growth rate, water intake (including spilled water), cleaning water usage, feed intake, feed composition, ambient temperature, slurry temperature, amount of inoculation and slurry storage time. MESPRO may be applied to pigs within the weight range of 20-110 kg. Slurry must also be collected beneath the piggery slats. MESPRO is a much more complex model than PigBal (DMDAMP and mass balance). It may be more accurate, but obtaining the input data is difficult. It is not a dynamic daily time-step model like MEDLI.

Gilbertson *et al.* (1979)

Gilbertson *et al.* (1979) developed a computer model that predicts the quantity and constituents of livestock manure from different management systems. The model allows for losses, gains and transfers of manure, calculating the total solids, volatile solids, N, P, K, Na, Ca, Mg, Fe, Zn, Mn, Cu, As and COD of the manure residue. ASAE and MWPS manure characteristics data was used as a base for their calculations. Animal weight and diet effects were assumed to be insignificant and all calculations were assumed to be independent. Losses through volatilisation of nitrogen and volatile solids were accounted for. When compared to previous research findings, the outputs of the model provided a reasonable approximation.

6.3.5. Evaluation of Predictive Models

Dougherty (1996) reviewed the relative merits of water, salt and nutrient (nitrogen and phosphorus) models. The models reviewed and the components they handle are summarised in Table 14.

TABLE 14 – LIST OF MODELS REVIEWED BY DOUGHERTY (1996).

Model	Authors/Reference	Water Balance	Salt balance	Nitrogen dynamics	Phosphorus dynamics
SWIM	Ross, 1984	Yes			
PERFECT	Littleboy *et al*., 1989	Yes		Limited	
SALF	Shaw and Thorburn, 1993		Yes		
Ryden and Pratt	Ryden and Pratt, 1984				Yes
Murtagh	Van Kuelen and Wolf, 1986	Yes			
LEACHM	Hutson and Wagenet, 1992	Yes	Yes	Yes	
EPIC	Sharpley and Williams, 1990	Yes			
AGNPS	Young *et al*., 1984	Yes		Yes	Yes
SPAW	Saxton et al., 1974	Yes			
MEDLI	QDPI, 1995	Yes	Yes	Yes	Yes
EPA	EPA, 1995	Yes			
CREAMS/GLEAMS	Knisel, 1980; Leonard *et al*., 1987	Yes		Yes	Yes
WATSUIT	Rhoades and Oster	Yes	Yes		
SODICS	Rose *et al*., 1979		Yes		

6.3.6. Predicting and Estimating Nutrient Excretion

McGahan *et al.* (2001) compared the nitrogen and phosphorus production from different classes of pigs using the universally adopted ASAE standards, the Australian publication 'Effluent at Work' and predictions from mass balance principles. Table 15 and Table 16 show the results for nitrogen and phosphorus respectively. There is a reasonable match between the data calculated by the mass balance and the data presented in "Effluent at Work". However, there is a poor correlation with the generally much higher results presented in the ASAE standards.

TABLE 15 – AMOUNT OF NITROGEN EXCRETED BY DIFFERENT PIG CLASSES (kg/yr)

Pig Class[1]	Mass Balance[2]	Effluent at Work	ASAE Standards
Gilts	12.0	11.7	24.7
Boars	15.0	15.0	33.2
Gestating Sows	13.9	11.7	37.0
Lactating Sows	27.1	-	37.0
Sucker	2.3	-	0.9
Sow + Litter		16.4	-
Weaner	3.9	2.9	3.1
Grower	9.2	7.7	7.6
Finisher	15.8	14.2	14.7

[1] Pig classes as per age and weight ranges described in McGahan *et al.* 2001.
[2] Mass balance is based on sorghum/wheat based diets, with 'typical' feed intake and feed wastage values as described by McGahan *et al.* 2001.

TABLE 16 – AMOUNT OF PHOSPHORUS EXCRETED BY DIFFERENT PIG CLASSES (kg/yr)

Pig Class[1]	Mass Balance[2]	Effluent at Work	ASAE Standards
Gilts	4.6	3.7	8.5
Boars	5.3	4.7	11.5
Gestating Sows	5.2	3.7	12.8
Lactating Sows	8.8		12.8
Sucker	0.4		0.3
Sow + Litter		4.0	
Weaner	1.1	1.1	1.1
Grower	3.0	2.6	2.6
Finisher	5.1	5.1	5.1

[1] Pig classes as per age and weight ranges described in McGahan *et al.* 2001.
[2] Mass balance is based on sorghum/wheat based diets, with 'typical' feed intake and feed wastage values as described by McGahan *et al.* 2001.

The salt content of a piggery effluent stream depends on the salt concentration of the feed and water used for drinking and cleaning. The amount can be estimated using mass balance principles. The final quality of the effluent available for irrigation is far more dynamic and is influenced by the percentage of water that is recycled for cleaning (under pen flushing), the amount of effluent irrigated, the climate and the surface area of the effluent treatment ponds. Table 10 shows that the measured salinity of piggery irrigation pond effluent ranges from 2.2 – 14.7 dS/m and Table 11 shows that measured salinity of piggery sludge ranges from 6.3 to 16.5 dS/m.

Within the piggery sheds themselves, and during pond-based treatment, there are significant losses of nitrogen from effluent via ammonia volatilisation. During pond treatment, nutrients are also partitioned to sludge. Details of these loss pathways and partitioning are discussed in the following sections.

6.3.7. Fate of Nitrogen After Excretion for Piggeries

Shed Losses

Significant nitrogen losses by ammonia volatilisation inside a piggery shed are likely. Vanderholm (1975) reported that up to fifty percent or more of nitrogen in fresh manure may be in the ammonia form or may be converted to the ammonia form very soon after excretion. Values for nitrogen loss through ammonia volatilisation in pig sheds is limited because of incomplete documentation of the total defecated, the severe restrictions of sample storage and analysis and the variability of field-testing (Overcash *et al*., 1983).

Values vary greatly depending upon the effluent collection system (flushing or static pits), and other factors such as pH, temperature and litter moisture content (Elliott and Collins, 1982). In a study of room temperature storage losses, Moore *et al.* (1983) found that Total Kjeldahl Nitrogen (TKN) losses from manure pits were less than ten percent after four weeks. TKN comprises the organic plus the ammonium nitrogen content. It is a more useful nitrogen measure compared to ammonia because even when nitrogen converts between the ammonia and organic forms the TKN remains constant (Overcash *et al*., 1983).

Overcash *et al.* (1983) showed that the ammonia (NH_3-N) fraction of TKN in piggery effluent varied considerably. They adopted an average value of 0.50 as the ratio of NH_3-N/TKN. Thus a considerable proportion of the total-N is available for loss immediately after excretion by pigs.

Using mass balance measurements of inputs and outputs from a section of a conventional, daily flush finisher shed, McGahan *et al.* (2001) predicted shed nitrogen losses of approximately 10%. Calculations by Casey (unpublished data) predicted shed losses of nitrogen via ammonia volatilisation of about 10% using measurements of ammonia concentration from these building types and predicted shed ventilation rates.

Studies have shown that for conventional flushed sheds in Australia, 10% of the nitrogen excreted by pigs is lost by ammonia volatilisation before the manure is removed from the building.

Partitioning to the Sludge

Howell (1976) reported that the initial nitrogen loss due to settling in a piggery effluent pond is approximately forty percent of the input manure. However, after sludge biological activity and transfer to the supernatant, the net amount of nitrogen deposited to the sludge is about fifteen percent.

A number of researchers have measured the nitrogen concentration in the sludge of anaerobic ponds receiving piggery effluent. Measurements from three investigators are provided in Table 17. Using mass balance principles, the fraction settled was also calculated.

TABLE 17 – CONCENTRATION OF TKN IN THE SLUDGE OF ANAEROBIC PONDS (FROM CASEY, 1992).

Study	Nitrogen Concentration (mg/l TKN)	Fraction Settled (Calculated)
Fullhage (1981)	4531	0.29
Booram *et al.* (1975)	3580	0.23
Barth and Kroes (1985)	2830	0.18
Average	**3647**	**0.23**

The PigBal spreadsheet and the MEDLI model use a “standard” nitrogen settling fraction of 23%. The amount that will settle depends on the treatment capacity of the pond and may vary between 15% and 30%.

Pond Losses

Humenik and Overcash (1976) proposed a dynamic model of nitrogen loss from an anaerobic pond using a mass balance approach. The major loss path for nitrogen from anaerobic ponds is surface volatilisation of ammonia nitrogen (Koelliker and Miner, 1973; Miller, 1976). The rate of ammonia volatilisation depends on the difference in the ammonia concentrations of the liquid and the air. Under field conditions the ammonia concentration in the air is quite low and the rate of loss is then directly related to the liquid concentration. Their model also shows that the rate of nitrogen loss is directly related to the surface area of the pond.

Typical nitrogen losses during pond treatment are quoted to be in the range of seventy to eighty percent (MWPS, 1985). Other data available has even observed losses as high as ninety-five percent and as low as fifty-three percent (Pano and Middlebrooks, 1982).

Typical nitrogen volatilisation rates predicted by the MEDLI model for piggeries range from 40% to 70%. Volatilisation rates depend on the pond surface area, climate and the ammonium concentration in the pond supernatant.

How much nitrogen is left after treatment

After accounting for nitrogen volatilisation losses from the piggery sheds and ponds (45% of the excreted nitrogen), PigBal estimates that 22% of the nitrogen excreted by a Standard Pig Unit remains in the sludge, with 33% of the excreted nitrogen remaining in the pond supernatant for irrigation.

6.3.8. Fate of Phosphorus After Excretion for Piggeries

Barnett (1994) measured the amount of total phosphorus and the proportion of four forms of phosphorus in the fresh, uncontaminated faeces of various livestock raised on commercial farms. In order of importance to plant growth, the four forms of phosphorus studied were:

- Inorganic.
- Residual – nucleic acid-type material.
- Acid-soluble organic – inositol hexaphosphates.
- Phospholipids – lipid.

Results collected from 16 pig finisher operations are provided in Table 18. Barnett (1994) compared his results to literature values and found that values had changed substantially from older results due to changes in animal production. Today, more attention is paid to quality of feedstuffs, nutritional balance, and supplementation to meet growth and maintenance requirements. In general he found that inorganic phosphorus constituted about half of the total phosphorus in fresh faeces.

TABLE 18 – TOTAL PHOSPHORUS, PROPORTION OF FOUR PHOSPHORUS FORMS AND DRY MATTER IN FRESH, UNCONTAMINATED FAECES OF 16 PIG FINISHER UNITS (BARNETT, 1994).

Form	Units	Mean	Range	Standard Deviation	Coefficient of Variance (%)
Total Phosphorus	g/kg	29.1	19.7 – 40.0	5.3	18.3
Inorganic Phosphorus	%	54.7	42.2 – 76.6	11.1	20.3
Residual Phosphorus	%	15.2	9.2 – 26.9	5.5	36.0
Acid-soluble Phosphorus	%	29.7	13.7 – 45.3	12.3	41.6
Lipid	%	0.4	0.3 – 0.5	0.1	15.8
Dry matter	g/kg	272.0	210 - 365	37.0	13.5

Phosphorus is not lost from the pond system through chemical or biological transformations. However, significant amounts are removed from the supernatant and concentrated in the sludge (Casey, 1992).

The phosphorus concentration in the sludge of anaerobic ponds receiving pig effluent has been measured by a number of researchers. Up to eighty percent of the phosphorus in the influent can accumulate in the sludge (MWPS 1985). Humenik and Overcash (1976) measured removal efficiencies for ortho-phosphorus of 90-94% for pilot ponds loaded at rates comparable with conventional anaerobic ponds.

The proportion of phosphorus partitioned to the sludge depends mainly on the treatment capacity of the pond and its hydraulic retention time. With the current Australian design standards for anaerobic pond treatment systems of at least 4 m^3 of pond capacity per pig, about 90% of total phosphorus entering the pond would most likely be partitioned to the sludge.

6.3.9. Fate of Salt in Piggeries

Potassium is usually found in anaerobic pond systems in its free cation form (Casey, 1992). Barth and Kroes (1985) measured similar potassium concentrations in both the supernatant and the sludge of piggery ponds. Overcash *et al.* (1978) studied the effect of pond pre-treatment on the removal of heavy metals and cations. They observed that there was little loss of potassium to the sludge. This also applies to other elements contributing to pond salinity such as chloride and sodium. Thus the electrical conductivity of the supernatant is likely to be similar to the in-situ sludge.

6.3.10. Fate and Prevalence of Heavy Metals in Piggeries

There is little measured data on the excretion of heavy metals by pigs. However, the amount that is excreted depends mainly on the diet of the pig. Both zinc and copper are elements that are commonly fed to weaner pigs to stimulate eating. Table 10 and Table 11 have some data on concentrations of copper, zinc and selenium in piggery effluent and solids.

6.3.11. Hydraulic Loading of Piggery Effluent Application

Due to the high nutrient concentration in piggery effluent (particularly nitrogen), nutrients will generally limit irrigation rates from piggeries before hydraulic loading.

However, to achieve the maximum possible dry matter production from a crop or pasture and hence the maximum removal of nutrients from the site, additional clean irrigation water will generally be required to meet the crop demand.

6.3.12. Standard Animal Unit and Nutrient Production Figures

Table 19 shows the Standard Pig Unit (SPU) multipliers for each pig class based on the estimated volatile solids production figures from Table 13.

TABLE 19 – STANDARD PIG UNIT (SPU) MULTIPLIER FOR EACH CLASS OF PIG

Pig Class	VS Production (kg/yr)	SPU Multiplier
Gilts	162	1.8
Boars	151	1.6
Gestating Sows	151	1.6
Lactating Sows	215	2.5
Suckers	11.0	0.1
Weaner pigs	47	0.5
Grower pigs	90	1.0
Finisher pigs	149	1.5

Using mass balance, “conservative” estimates of nutrients (N and P) available for utilisation can be developed (Table 20), whether the pigs are housed in conventional sheds or deep litter systems. These figures are derived from the information contained in Sections 6.3.7 and 6.3.8.

TABLE 20 – PREDICTED NITROGEN AND PHOSPHORUS MASSES PRODUCED BY A 1000 SPU PIGGERY

	Nitrogen (kg/yr)	P (kg/yr)
Conventional Shed		
Amount excreted	9200	3000
% Lost in the shed	10	0
Amount out of shed	8280	3000
% Lost during pond treatment	40	0
Amount left in ponds	5000	3000
% Lost during and after application*	20	
Amount left for plant uptake	**4000**	**3000**
Deep Litter Shed		
Amount excreted	9200	3000
% Lost in the shed	10	0
Amount out of shed	8280	3000
% Lost during storage/composting	25	0
Amount out of storage	6210	3000
% Lost during and after application	20	0
Amount left for plant uptake	**5000**	**3000**

* Refer to section 8.1.2 for estimation on nitrogen volatilisation losses for irrigation. If irrigation is performed predominantly at night-time, nitrogen volatilisation are likely to be only 5 – 10%.

The figures in Table 20 provide a surrogate measure for calculations or modelling. For conventional sheds it is assumed that all of the sludge stored in the pond also needs to be applied on-site. These figures are only likely to be useful for operations with large utilisation areas on-site.

6.4. Estimating Nutrients and Salts in Feedlot Cattle Effluent & Manure – Concentration and Quantity Method

For an operating feedlot, the concentration of elements in the effluent and manure, and the quantity of effluent and manure provide a measure of the mass of nutrient for reuse.

The measured characteristics of feedlot effluent and solids have been collated from a number of sources. The "Designing Better Feedlots" data was collected in the early 1990's, with some more recent data collected by the DPI Queensland from 11 feedlots in Southern Queensland. Table 21 provides more recently measured data showing substantially lower nutrient concentrations in the retention pond effluent. The average total nitrogen concentration has dropped almost four-fold (720 mg/L to 190 mg/L). The average phosphorus concentration has almost halved (104 mg/L to 65 mg/L). This is probably due mainly to the increased pen cleaning now practised in Australian feedlots. This reduces the amount of nutrients exported in the runoff and increases the amount transferred to the manure stockpile. Table 22 shows typically measured concentrations of various elements in stockpiled feedlot manure. These results show a wide variation in the reported data. Thus "typical or average" pond supernatant (irrigation water) and stockpiled manure concentrations of nutrients and salts cannot be provided. This is because of the wide variation of design, management, diets and climatic conditions.

Methods for measuring the quantity of effluent or manure are provided in Sections 7.1.2 and 7.1.4.

TABLE 21 – CHARACTERISTICS OF FEEDLOT POND EFFLUENT, AVERAGE AND (RANGE)

		Designing Better Feedlots[1]	DPI Qld (2001) – Unpublished data[2]
Dry matter	% w.b.	1.57 (1.2 – 2.6)	
Volatile solids	% d.b	48.56 (39.23 – 62)	
Ash	% d.b	51.44 (38 – 60.77)	
pH		7.43 (6.9 – 8.1)	8.0 (7.2 – 9.1)
COD	mg/l	9579.2 (4862 – 16806)	
Total Nitrogen	mg/l	720.55 (286 – 1155)	
TKN			188 (46 – 333)
Ammonium Nitrogen	mg/l		139 (37 – 277)
Total Phosphorus	mg/l	103.76 (26 – 440)	65 (22 – 114)
Ortho-Phosphorus	mg/l		20 (7 – 45)
Potassium	mg/l	2370 (985 – 9102)	784 (307 – 2800)
Sulphate	mg/l		59 (1 –317)
Boron	mg/l		
Kjeldahl Copper	mg/l		0.100 (0.025 – 0.187)
Dissolved Iron	mg/l		1.45 (0.40 – 4.80)
Manganese	mg/l		0.18 (0.05 – 0.54)
Zinc	mg/l		0.40 (0.05 – 1.03)
Calcium	mg/l		65 (25 – 118)
Magnesium	mg/l		158 (59 –441)
Sodium	mg/l		473 (102 – 933)
Chloride	mg/l	420 (333 – 674)	1256 (370 – 2660)
Conductivity	dS/m	13.19 (3.88 – 37.8)	6.8 (2.2 – 11.4)
SAR			7.15 (2.2 – 14.5)

[1]. Designing Better Feedlots - Data from ASAE, Powell and DPI
[2]. DPI Qld 2001 – 11 Feedlots on the Darling Downs

TABLE 22 – CHARACTERISTICS OF STOCKPILED FEEDLOT MANURE, AVERAGE AND (RANGE)

Component	Units	Average and (Range)*
Dry matter	% w.b.	72.97 (53.7 - 92)
Volatile solids	% d.b	67.6 (55 - 75.9)
Ash	% d.b	32.4 (24.1 - 45)
pH		6.95 (5.6 - 9.2)
Total Nitrogen	% d.b	2.18 (1 – 3)
Ammonium Nitrogen	% d.b	0.038 (0.036 – 0.169)
Total Phosphorus	% d.b	0.8 (0.4 – 1.3)
Potassium	% d.b	2.32 (1.5 – 4.0)
Sodium	% d.b	0.61 (0.3 – 1.3)
Chloride	% d.b	1.35 (0.7 – 2.3)
Conductivity	dS/m	12.36 (3.9 – 22)
SAR		5.9 (0.8 – 18.8)

*Skerman (2000) and Gardner *et al.* (1994) - interpreted from Powell (1994).

6.5. Estimating Nutrients and Salts in Feedlot Cattle Effluent & Manure – Emissions Data

Providing a generic emission factor for feedlot cattle is difficult because reported values for feedlot cattle excretion vary widely for total solids (TS), volatile solids (VS), nitrogen (N) and phosphorus (P). Partly, this is because the literature is generally based on manure production estimated from animal mass and does not consider likely manure production based on feed intake. Table 23 shows comparisons of manure production by TS, VS, N and P for a 600 kg beef animal, from four different references (Van Sliedregt *et al*., 2000).

TABLE 23 – COMPARISON OF SOLIDS AND NUTRIENT PRODUCTION FOR A 600kg LIVEWEIGHT BEEF ANIMAL (VAN SLIEDREGT *ET AL*., 2000)

Manure Component	ASAE[1] (1998)	MWPS[2] (1985)	USDA-SCS[3] (1992)	Designing Better Feedlots (Watts et al., 1994)[4]
Volatile Solids	1576.8	1576.8	1191.4	1105.0
Total Solids	1861.5	1857.1	1294.3	1300.0
Nitrogen	74.5	75.3	65.7	76.7
Phosphorus	20.1	54.3	20.6	20.8

[1] ASAE – American Society of Agricultural Engineers (Standards).
[2] MWPS – Mid West Planning Service (Livestock Waste Facilities Handbook).
[3] USDA –SCS – United States Department of Agriculture – Soil Conservation Service (Agricultural Waste Management Fieldbook).
[4] Designing Better Feedlots – Characteristics of Feedlot Wastes pp. 7.5-7.15

The Reference Manual for the Establishment and Operation of Beef Cattle Feedlots in Queensland defines the carrying capacity of feedlots by the number of Standard Cattle Units (SCU). A SCU is defined as an animal of 600 kg liveweight, at the time of exit (turn off) from the feedlot (Skerman, 2000).

The term SCU was first published in the Queensland Stock Act – Cattle Feedlot Regulations (1989). It presumes manure production is a function of animal mass. Literature available when this concept was developed indicated that feed intake was related to liveweight, or metabolic weight (NRC, 1984). The SCU aimed to provide a consistent basis for evaluating feedlots for environmental purposes.

Introduction of the SCU concept enabled the stocking capacity of feedlots to be expressed in accordance with the weight of cattle turned off from the facility, rather than the number of head. It was later adopted in other states of Australia. Conversion factors were derived, based on the metabolic weight of an animal (*ie.* liveweight$^{0.75}$). For example, the SCU for an animal of 400 kg liveweight at turn-off is calculated as $400^{0.75}/600^{0.75}$ or 0.74 SCU.

Sinclair (1997) questioned the validity of the SCU concept. Since his study did not show increasing dry matter intakes with increasing liveweight (SCU), he also questioned the value of SCU for estimating manure production. He suggested that SCU (viz liveweight) did not influence manure production within the live weight range of 240 kg to 377 kg. Van Horn et al., (1994) and Morse et al., (1994) also report no direct relationship between manure production and animal weight. After reviewing data, Van Horn (1992) concluded estimates from the dietary intake of a nutrient, minus amount secreted in milk provided a good prediction of total mineral excretion by mature dairy cows and one on which to base manure management systems.

6.6. Estimating Nutrients and Salts in Feedlot Cattle Effluent & Manure – Mass Balance Principles

This section provides details to estimate the quantity of nutrients and salts in cattle feedlot effluent and solids using mass balance principles. Information on predictive models that are based on mass balance principles is also included.

6.6.1. Mass Balance Principles for Cattle Feedlots

In the last ten years, the expertise of the feedlot industry and the specialist feeding of animals for specific markets has developed significantly. There has also been concurrent extensive research into animal growth and composition, the factors influencing feed intake and digestibility, feed composition and manure and effluent management. This has allowed for better predictions of manure output from feedlot cattle using mass balance principles.

As with piggeries, the TS, VS, and nutrient (N and P) content of the manure is the most appropriate system to measure the potential environmental impact of a feedlot. Van Sliedregt *et al.* (2000) suggested that a more accurate model for predicting feedlot cattle excretion should be based on feed intake, feed digestibility and mass balance principles. These principles estimate the quantity of nutrients and salt in effluent and manure as the difference between inputs (cattle, feed and water) and outputs (cattle and nitrogen volatilisation). This was achieved by upgrading the BeefBal model.

Skerman (2000) reports that the salinity of pond effluent and feedlot manure is variable and apparently closely related to the salinity of the cattle drinking water and the salt content of the diet. As with nitrogen and phosphorus, the components of the salt in feedlot effluent and manure (sodium, chloride, magnesium, calcium etc) can be traced using mass balance principles. If the cattle consume drinking water with a high salt content or are fed a diet containing a significant sodium chloride content, the salinity of the effluent and manure increases. Due to the highly variable of salt levels in feedlot cattle drinking water it is difficult to provide a typical value for the salt content of excreted feedlot manure (urine + faeces).

Gardner *et al.* (1994) report that all feeds contain inorganic salts in concentrations reflecting their uptake by the growing plant. These elements are associated with, or are bound up in organic plant compounds. Thus they are assumed to be excreted by the animal primarily in this form and hence contribute little to the pool of water-soluble inorganic salts.

6.6.2. BeefBal

BeefBal (Watts *et al.,* 1994) is a Microsoft Excel© based spreadsheet model developed to estimate the quality and quantity of manure produced by cattle feedlots, and to assess the environmental sustainability of associated reuse practices.

BeefBal provides a mass balance of the nitrogen, phosphorus, potassium and salt entering the feedlot system (via incoming cattle, feed and drinking water) to determine the masses of nutrients and salt in the manure and liquid effluent produced by the feedlot. The model then uses this manure output data to assess the sustainability of the associated reuse areas. In assessing application rates, the model considers the nutrient uptake of the crop, the nutrient storage capacity of the soil and the expected nutrient losses to the environment (gaseous losses).

Recent enhancements to the model include the incorporation of a DAMP-based (digestibility approximation of manure production) method to determine the "as excreted" manure constituents, using a wide range of possible ration ingredients and cattle classes (eg. Jap Ox, Korean, Domestic Trade). Predictions of the amount of TS, VS, nitrogen and phosphorus for different classes of feedlot cattle are shown in Table 24.

TABLE 24 – THE PREDICTION OF FEEDLOT MANURE FOR THE DIFFERENT CLASSES OF ANIMAL FROM THE DMDAMP MODEL USING A BARLEY BASED DIET (VAN SLIEDREGT ET AL, 2000).

Class of animal	Domestic	Domestic	Korean	Jap-Ox	Jap-Ox	Jap-Ox
Days on feed	70	100	150	200	250	300
Sorghum						
TS excreted (kg/yr)	1023	1087	1204	1220	1203	1193
VS excreted (kg/yr)	745	794	894	901	887	877
N excreted (kg/yr)	65.0	68.5	73.0	76.7	76.6	76.5
P excreted (kg/yr)	9.1	9.7	10.5	11.2	11.3	11.3

The model has not yet been developed to a fully commercial standard. It is currently used primarily for internal DPI, Queensland research and assessment. However, a number of consultants use the model (or its derivatives) to assist in preparing applications for new and expanding feedlots and for developing effluent and manure management strategies for existing feedlots. Copies of the model are available upon request from the DPI, Queensland, on the understanding that the model has not been finalised as not all outputs have been thoroughly validated against measured data from operational feedlots.

6.6.3. MEDLI

A detailed description of MEDLI can be found in Section 6.3.3.

Blair and E.A. Systems (2002) suggest that the MEDLI model is the most 'useful' tool for modelling effluent irrigation for cattle feedlots because it is structured specifically to consider effluent irrigation. However, the model is inflexible since it does not consider the usual management inputs to reuse areas (eg. changes in cropping regimes such as the use of rotation, addition of manures, inorganic fertilisers or ameliorants such as lime or gypsum). Therefore it provides a conservative result. However, it can also provide a worst case outcome because no allowance is made for management changes to the soil-crop system (as would occur in real practice) in response to adverse changes.

MEDLI has not been extensively used to design effluent reuse areas for feedlots. However, it has been found to be a very useful tool for performing water balances with feedlots. MEDLI's predictions of the nutrient concentration in the effluent irrigation water appear to be lower than those suggested by historical data. However recently collected data from modern, well maintained feedlots shows that effluent concentrations are much lower and closer to the predictions of MEDLI.

6.6.4. Other Predictive Models

Refer to section 6.3.4 for a review of the water, salt, nitrogen and phosphorus estimation model reviewed by Dougherty (1996).

There are no other known commercially available models for predicting the manure and effluent produced by cattle feedlots in Australia.

6.6.5. Fate of Nitrogen in Feedlots

Nitrogen can be lost from the feedlot system via ammonia volatilisation and denitrification. Consequently, it is difficult to calculate a nitrogen mass balance and partition nitrogen between the pond effluent and the solid manure stockpile.

Nitrogen excreted by cattle is in both organic and inorganic forms. The organic forms are primarily contained in the faeces. They are either readily available for conversion to the inorganic ammonium form by soil micro-organisms (mineralisation) or slowly available to the mineralisation process. In the faeces, undigested proteins and unabsorbed sugars are readily available for decomposition whilst structural carbohydrates in plant material are slowly available (Gardner *et al.*, 1994).

The other major pathway of nitrogen excretion is via the urine. About 70 % of the nitrogen in urine is urea and about 30% is readily mineralised organic compounds. Urea readily breaks down when exposed to moisture, air, and the urease enzyme, producing aqueous ammonium. Aqueous ammonium (NH_4^+) is available for retention by cation exchange sites on soils (when spread) or for conversion into aqueous ammonia (NH_3), which is potentially available for gaseous loss to the atmosphere (ammonia volatilisation) (Gardner *et al.*, 1994).

Gardner *et al.* (1994) assume that 60% of the nitrogen excreted is in the urine, and that all of this nitrogen is lost by volatilisation because:

- Pen manure has a poor pH buffering capacity
- Pen manure has a low cation exchange capacity
- There would be a warm temperature below closely spaced cattle that are good radiators of heat

Gardner *et al.* (1994) assumed that there is no loss of nitrogen due to denitrification because the feedlot pad is an aerobic environment. This assumption is supported by analysis of manure of various ages that shows that ammonium is the major inorganic nitrogen form. This suggests that little nitrification occurs on the pen surface or in the manure stockpile. Similarly, for well-constructed and maintained pen surfaces there should be little drainage below the manure-soil interface.

During rainfall events, nitrogen in eroded manure is transported via the settling basin to the effluent pond. Any nitrogen deposited in the settling basin eventually returns to the manure stockpile.

The amount of nitrogen entering the effluent pond depends on the surface hydraulics of the feedlot, the settling efficiency of the sedimentation basin and the size distribution (and hence settling time) of the eroded manure. Once in the pond the manure particles settle to the bottom as sludge, remain in suspension in the supernatant or are transformed to the inorganic ammonium form during anaerobic fermentation (Gardner *et al.,* 1994). Any manure settling to the sludge layer is ultimately returned to the manure stockpile.

The mineralised ammonium is additional to the water-soluble nitrate and ammonia transported in the runoff water. Once in the ammonium form, the nitrogen is again available for volatilisation at a rate depending on the pond pH and temperature, wind speed and its surface area to volume ratio (Gardner *et al.* 1994).

There is little experimental data on the volatilisation rates of ammonium from feedlot ponds. However, like piggery ponds the volatilisation rates are likely to be around 50%, with ranges from 20 – 80%.

If nitrogen were a conservative element like phosphorus, its concentration in the manure would increase with dry matter losses as the manure ages. However, this is not the case and nitrogen content decreases with increasing manure age, suggesting nitrogen losses exceed the dry matter loss rates (decomposition). Mass balance calculations suggest that the nitrogen loss from the stockpile is around 40-50%.

Due to the numerous nitrogen loss pathways, the most accurate method for determining the nitrogen content of feedlot manure and effluent is by measurement.

Feedlot manure sampled from manure stockpiles contains an average of about 2.2% total nitrogen. Current data suggests that over 90% of the total nitrogen is in the organic form, while the remainder is in the inorganic ammonium-nitrogen or nitrate-nitrogen forms. Ammonium-nitrogen levels are generally less than 5% of the total nitrogen.

Predictions of nitrogen loss rates from feedlot pads and manure stockpiles using BeefBal indicate that 22% of the nitrogen excreted by a Standard Cattle Unit remains in the manure after stockpiling, with a further 3% of the excreted nitrogen being exported to the settling basin and retention pond in runoff.

6.6.6. Fate of Phosphorus in Feedlots

The measurement of total phosphorus and the proportion of four forms of phosphorus in the fresh, uncontaminated faeces of various livestock by Barnett (1994) was described in section 6.3.8. The data collected from nine herds of finisher cattle is shown in Table 25.

TABLE 25 – TOTAL PHOSPHORUS, PROPORTION OF FOUR PHOSPHORUS FORMS AND DRY MATTER IN FRESH, UNCONTAMINATED FAECES OF NINE FINISHER HERDS OF CATTLE (BARNETT, 1994).

Form	Units	Mean	Range	Standard Deviation	Coefficient of Variance (%)
Total Phosphorus	g/kg	6.7	3.7 – 10.6	2.1	32.0
Inorganic Phosphorus	%	48.3	28.2 – 63.2	12.1	25.1
Residual Phosphorus	%	40.8	20.7 – 56.4	11.3	27.8
Acid-soluble Phosphorus	%	8.9	3.0 – 14.9	4.0	44.7
Lipid	%	1.9	1.5 – 3.3	0.7	34.3
Dry matter	g/kg	172.0	136 - 241	37.0	21.8

BeefBal predictions for the partitioning of phosphorus to the manure stockpile and retention pond indicate that 97% of the phosphorus excreted by a Standard Cattle Unit remains in the manure for stockpiling, with the remaining 3% of the excreted phosphorus being exported to the settling basin and retention pond in runoff.

6.6.7. Fate of Salts in Feedlots

Salt enters a feedlot via the drinking water, diet supplements, diet and rainfall. It is difficult to predict salt partitioning between the solid manure and the feedlot pond. Gardner *et al.* (1994) suggested two possible approaches:

- Assume a certain fraction of excreted salt is exported to the pond via pen runoff and then calculate the pond salinity. The residual salt can be used to calculate manure salinity levels which can be compared with measured values.
- Measure manure salinity levels and calculate salt retained in the manure. The difference from total salt in is salt exported to the pond.

Using this second approach, Gardner *et al.* (1994) predicted that salt exported to the pond is over 80% of the salt excreted to the pen surface.

The salinity of the pond effluent fluctuates with the water balance of the pond, increasing during drier periods (due to evaporation) and decreasing during wet periods. These changes in pond salinity are best predicted using mass balance principles or computer models, such as MEDLI.

6.6.8. Fate and Prevalence of Heavy Metal in Cattle Feedlots

The manure produced by cattle feedlots is nutrient rich and usually contains low levels of heavy metals and other contaminants. As testing for heavy metals in cattle feedlot manure is uncommon, limited test data is available, and data that is available is often drawn from a limited number of samples. Measured copper and zinc concentrations vary from very low to above threshold values. Measured data from stockpiled manure from Queensland feedlots showed that copper concentrations averaged 30 mg/kg (range 14–71 mg/kg) and zinc concentrations averaged 154 mg/kg (range 80–283 mg/kg).

6.6.9. Standard Animal Unit and Nutrient Production Figures

Table 26 shows modified Standard Cattle Unit (SCU) multipliers for each feedlot animal class based on the estimated volatile solids production figures from Table 24. These multipliers assume that one SCU is equivalent to a Korean steer (150 days on feed, liveweight in of 380 kg, and liveweight out of 600 kg).

TABLE 26 – MODIFIED STANDARD CATTLE UNIT (SCU) MULTIPLIER FOR EACH CLASS OF FEEDLOT ANIMAL USING VS PRODUCTION (BARLEY DIET).

Feedlot Animal Class	VS Production (kg/yr)	SCU Multiplier
Domestic – 70 days	745	0.83
Domestic – 100 days	794	0.89
Korean – 150 days	894	1.00
Jap – Ox – 200 days	901	1.01
Jap Ox – 250 days	887	0.99
Jap Ox – 30 days	877	0.98

Using mass balance, "conservative" numbers can be developed for the mass of nutrients (nitrogen and phosphorus) remaining for utilisation from 1000 SCU (Table 27). These figures are derived from the information contained in Sections 6.6.5 and 6.6.6.

TABLE 27 – PREDICTED NITROGEN AND PHOSPHORUS AVAILABLE FROM 1000 SCU

	Nitrogen (kg/yr)	Phosphorus (kg/yr)
Amount excreted	77,000	11,300
% Lost on pad shed	60	0
Amount left on pad	46,200	11,300
% Lost during pond and stockpile	40	0
Amount left for application	18,500	11,300
% Lost during and after application	20	
Amount left for plant uptake	**15,000**	**11,300**

The figures in Table 27 can be used in the absence of any site specific mass balance calculations or modelling. They assume that all of the nutrients contained in the stockpiled solids, pond effluent and pond sludge will be applied on-site. These figures are only likely to be useful for operations having large utilisation areas on-site.

6.7. Risk Assessment: Nutrients in Manure and Effluent

To be able to manage a reuse area in an environmentally sustainable manner it is important to know the amount of nutrients applied. Better quantification of the nutrients for reuse, allows for better management. This section provides a process for assessing the risk an enterprise poses to the environment based on the level of precision of nutrient quantification. It is important to note that quantification or estimation of the nutrients for reuse is only a small part of their sustainable reuse. The vulnerability of natural resources (surface water, groundwater and soil) and the overall standard of design and management are also important. These are considered further in Section 11.2.

Low Risk	The quantity of effluent and solids reused is measured and the quality of effluent and solids reused is regularly measured (at least annually, more frequently if required to ensure sound management of nutrients). OR You have developed a mass balance of nutrient production from your piggery or cattle feedlot using accepted design tools, such as PigBal, BeefBal or MEDLI using conservative figures. (There can be a great variation in nutrient predictions from mass balance models).
High Risk	You have never measured, but only estimated the mass of nutrients applied using "text-book" values, such as those provided in Table 20 for piggeries and Table 27 for cattle feedlots.

A risk weighting of 1 or 3 applies to the Nutrients and Manure criterion. A low risk attracts a risk weighting of "1" and high risk attracts a risk weighting of "3".

These numbers are transferred to Table 54, Table 55 and Table 56 in Section 11.2.

7. DESIGN & MANAGEMENT OF REUSE AREAS

The design and management of reuse areas are important in deciding the risk of environmental impacts, particularly where vulnerable resources are concerned. This section provides background information on design and management options. The design and management practices partly determine the environmental risk of a reuse activity. They feed into the risk assessment process, along with the predictions of nutrients produced (mass balance), the assessment of the natural resources and the sustainability indicators.

The end of this section includes a process for evaluating the environmental risk associated with the design and management of the reuse areas.

7.1. Land Areas Available and Application Methods

7.1.1. Size of Land Area

The area of land needed for sustainable reuse of effluent or manure depends on the land use and crop yield since the quantity of nutrients usually limits the application rate for piggery and cattle feedlot by-products.

The mass of nitrogen and phosphorus supplied by effluent irrigation or by spreading solid by-products should not exceed the level removed through plant harvest or grazing stock liveweight gain, plus soil storage (phosphorus) and volatilisation losses (nitrogen). Guidelines for nutrient removal rates by various land uses are provided in Section 7.3. Grazing removes only very small nutrient masses and is not a preferred land use option.

The mass of nutrients applied by irrigation (kg/ha/yr) equals the irrigation rate (ML/ha/yr) multiplied by the nutrient content of the effluent (mg/L which is the same as kg/ML). To apply the target mass of nutrients, reliable methods for determining the effluent application rate and the time period of effluent application are needed. The nutrient concentration of effluent varies depending on the source of effluent, type of piggery or feedlot cattle operation, design of effluent treatment ponds, climate and irrigation practices. It is important to periodically analyse the effluent so that the amount of nutrients applied by irrigation can be quantified. These can then be matched to expected nutrient uptake by crops or liveweight gain, plus volatilisation losses and phosphorus storage.

Nitrogen volatilisation rates for irrigation vary depending on the application method, the percentage of nitrogen that is in the ammonium form, the pH of the effluent and when it is applied (day or night). For effluent irrigation, spray methods may allow 20% of the nitrogen to be lost by volatilisation. Less nitrogen (say 10%) is lost when surface flow methods are used. Volatilisation rates for non-composted piggery separated solids and feedlot manure could be quite high (about 30%). However, volatilisation rates from sludges, spent bedding and composted material are likely to be considerably lower (say 10%).

Typical phosphorus sorption rates for different soil types are presented in Table 33.

Due to the high concentration of nutrients in pond treated piggery and cattle feedlot effluent, the required reuse area will probably be limited by the crop-soil processes responsible for assimilation of the nutrients in the effluent. Hydraulic loading is very rarely the limiting factor when sizing a reuse area solely for effluent reuse. However, adding clean water to dilute the effluent increases the total irrigation volume. This helps to meet the water demand of the crop, but may also mean that hydraulic loading may become an issue needing careful management.

7.1.2. Measuring Effluent Applied

There are numerous methods for calculating the volume of effluent applied during an irrigation event.

A flow meter can accurately measure the effluent flow rate during irrigation. Multiplying the flow rate by the time of irrigation calculates the applied volume. Several types of flow meters are available. If in-line flow meters are used, non-corrosive types should be selected. Alternatively, non-contact ultra-sonic, doppler and non-contact magnetic flow meters are available. However these are typically very expensive.

A depth gauge in the pond, used with a storage capacity curve, provides an estimate of the irrigation rate when large volumes are irrigated at a time. The curve shows the volume of effluent in the pond when filled to any depth. The change in depth from the start to the finish of the irrigation should be measured.

For a single hand-shift type sprinkler, the pumping rate can be estimated from the time taken to fill a container of known volume. The flow rate must be measured from the irrigation nozzle. It can be very difficult to measure effluent volumes this way. A plastic hose fitted over the nozzle and directed to a 10 L bucket helps. For a sprayline, the outflow from at least three nozzles should be measured. Both sides of double-sided nozzles should be measured. Provided there are not too many pipe-join leaks, this method gives a good estimation.

If effluent is pumped from a tank or sump of known capacity, daily or weekly irrigation volumes may be estimated from the sump or tank volume and the emptying frequency.

If bulk tankers are used to spread effluent, tanker volume and emptying frequency provide a good estimate of the irrigation rate.

The volume of effluent released should be recorded each time effluent is irrigated, along with the paddock to which the effluent is applied.

7.1.3. Effluent Dispersion Methods

A range of effluent dispersion methods is available. The most common methods are spray or flood irrigation. However, trickle or drip irrigation and tanker spreading may sometimes be applicable. Because nutrient loads, and not hydraulic loads limit target application rates, irrigators must be able to apply effluent at much lower rates than water irrigation.

Spray Irrigation Systems

Spray irrigation systems pump effluent through pipelines for discharge through sprinklers under pressure. Spray systems suit most soil types and a range of slopes (up to 10%). They distribute effluent evenly, allowing for good control over application rates and minimal runoff. However, distribution may be distorted by the wind, creating potential for odour and spray drift. Spray systems should not be used to irrigate human food crops that are eaten raw due to potential disease risk. In all spray systems, nozzle clogging by solids or mineral accumulations can occur.

Hand-shift irrigation systems are a flexible, low-cost option. However, they are labour-intensive and therefore only suit areas of up to 10 ha.

Fixed-sprinkler irrigators have a higher capital cost than hand-shift systems, but a much lower labour requirement. These only work well with very clean effluent.

Small travelling irrigators suit low flow rates and can apply light applications of effluent. They are best used on small to medium sized effluent irrigation areas (up to 20 ha). They have relatively low capital and operating costs.

Lateral move and centre pivot irrigators have high capital and operating costs. They are very effective for large areas. They suit land with a uniform slope of up to 5%. Lateral move irrigators can only irrigate rectangular areas.

High-pressure systems such as travelling big guns create very fine aerosols and thus application areas need to be carefully designed and managed if these systems are used close to receptors.

Flood Irrigation Systems

Flood irrigation systems release large volumes of water over a defined land area. Slope, banks and sometimes furrows, control the movement of the water. Terminal ponds at the base of the irrigation areas catch runoff water. These systems disperse nutrients unevenly unless the land slope is very even and the soil heavy. The effluent passes too quickly through sandy soils posing a risk to groundwater contamination. If used on duplex soils the effluent passes rapidly through the light topsoil and can then move laterally over the heavier subsoil posing a risk of groundwater and surface water contamination.

Work by Redding (2002) with piggery effluent shows the phosphorus can leach through the soil profile via bypass flow, which is most likely to occur when the surface of the soil becomes saturated. This work highlights the need to avoid using effluent irrigation techniques that promote surface soil saturation, especially methods like flood and contour irrigation.

Drip Irrigation Systems

Drip irrigation systems usually comprise a network of small diameter pipes delivering effluent directly to the soil near plants. These systems best suit tree crops and some horticultural row crops. They can be used to apply effluent to most human food crops except where the fruit touches the ground. Because these systems clog easily they are only suitable for irrigating well-filtered effluent. There are also travelling drip irrigators.

Tanker Spreading

Tanker spreading involves filling a vacuum tanker with effluent and spreading the effluent using a spray nozzle and deflector plate, surface drop pipes or sub-surface injection behind tines. Tanker spreading is highly labour intensive, and machinery operating costs can be high particularly if directly injecting effluent. Tanker spreading is best suited to spreading slurries of 5-10% solids and for transporting over short distances (e.g. < 10 kms).

7.1.4. Measuring Solids Applied

The spreading rate for solid by-products must achieve a match between the nitrogen and phosphorus mass added and the mass removed by harvesting plants or liveweight gain from the spreading areas, plus nitrogen volatilisation losses and phosphorus soil storage. Guidelines for nutrient removal rates by various land uses are provided in Section 7.3.

To calculate the spreading rate matching the target nutrient application rate, the nitrogen and phosphorus concentration of the solids need to be determined (mg/L or mg/kg). The salt content of the sludge should also be determined, particularly if effluent is recycled as flushing water.

At each spreading it is important to record: the quantity of solids spread, the paddock involved, the solids spreading rate (m^3/ha or kg/ha) and the mass of nitrogen and phosphorus applied (calculated from analysis data).

7.1.5. Solids Spreading Methods

Wet solids are usually spread using a vacuum tanker. Dry solids are usually spread using a manure spreader or a fertiliser spreader. It is important that the method chosen achieves uniform spreading at sustainable rates.

7.2. Using Effluent and Solid By-Products to Improve Soil Structure

This section describes some of the benefits to soil that result from the reuse of intensive animal by-products. This includes the reuse of solids with effluent to overcome difficulties associated with using highly saline effluent.

7.2.1. Applying Effluent with Solids

Effluent from both piggeries and feedlots can have a high sodium content. Since plants require very little sodium to grow, this can lead to sodium accumulation in the soil profile and associated soil degradation through salinity and sodicity. Accumulating salts in the soil profile are leached through the soil profile with rainfall and irrigation. However, the salt leaching rate depends upon the quality of the effluent, the soil type and water additions (rainfall and irrigation). High salt leaching rates may also be associated with high leaching rates for other compounds e.g. nitrate-nitrogen. The addition of solid by-products containing significant organic matter helps alleviate some of the deleterious effects of saline effluent by improving soil structure. The application of both effluents and solids can be advantageous to crop production and hence sustainability, providing the nutrient content of both is added to match the removal by the crop.

7.2.2. Improving Soils

Feedlot manure has several therapeutic effects on soil, helping to improve soil structure, porosity and water infiltration rate, and increasing soil water holding capacity. In the latter case, the increase in infiltration reduces runoff, increases the potential for soil water storage, and thus increases the potential for plant productivity through a greater supply of water (Lott, 2000). These similar advantages can be obtained from the application of piggery solid by-products.

Organic matter promotes healthy soil structure by forming organic clay bonds. It improves the cation exchange capacity (CEC) of sandy soils and provides a nutrient source. Organic carbon is an indirect measure of organic matter. The organic carbon concentration can be approximately converted to an organic matter concentration by multiplying by 1.75.

The organic carbon concentration of undisturbed soil is typically 1 – 3%, although this varies with soil type. Continuous cultivation tends to reduce organic carbon levels and causes soil structure decline. Piggery and feedlot effluent and solid by-products both contain significant quantities of organic matter. Adding these helps to raise soil organic carbon levels.

7.2.3. The Difference Between Composted and Raw Manure

A discussion on the relative merits of compost versus raw manure is contained in the Biocycle Magazine (Anon, 2002). Very little happens to phosphorus during the composting of manure. The total phosphorus will be preserved during the process, with only small amounts lost through erosion with runoff or leachate. The form of phosphorus also does not change much during composting. Several research projects have shown that the phosphorus in manure and the phosphorus in the resulting compost are both equally water soluble and available to plants. On the other hand, at least one research project suggests that, on a percentage basis the phosphorus in composted poultry manure is slightly less water-soluble than phosphorus in the original manure. The reduction in available phosphorus probably represents the phosphorus metabolised and retained by microorganisms in decomposing the manure. If there is a difference in the availability of phosphorus in manure versus compost, it is probably small.

During the composting process, much of the organic matter is lost, which concentrates the phosphorus in the final product. A considerable amount of the nitrogen is also lost via ammonia volatilisation. This will in turn reduce the ratio of nitrogen to phosphorus, so the compost contains more phosphorus per unit of nitrogen compared to raw manure. As most solid manure by-products already have an excess of phosphorus compared to nitrogen to meet plant requirements, composting further affects this ratio. However, this effect is likely to be negated, as the nitrogen in manure is still relatively unstable and mobile. That is, during and after the application of raw manure, a significant amount of the nitrogen will volatilise as ammonia. Due to this loss of nitrogen, the ratio of nitrogen to phosphorus in the remaining manure can fall to the same level as the compost.

The relative merits of whether to compost manure or not will depend on the market for the manure. Factors to consider include distance, pathogens, nitrogen content and odour impacts.

7.3. Type of Crop/Pasture Grown and Yield

The type of crop grown on the reuse area determines the nutrient uptake through its dry matter yield and nutrient content. Table 28 shows typical dry matter nutrient contents and expected yield ranges for a variety of pasture, silage, hay, grain and horticultural crops. The yields presented are for typical cropping soils. Further information for other crops can be found in various references, such as the Draft Guidelines of the "Use of Effluent in Irrigation", prepared by the NSW EPA.

For example, a maize silage crop with a dry matter yield of 10 t/ha/yr contains 300 kg of nitrogen, 50 kg of phosphorus and 200 kg of potassium.

Grazed pasture is an ineffective method of utilising nutrients from reuse areas. Most of the nutrients are simply recycled through the grazing animal and returned to the reuse area. Grazing systems typically require at least ten times more area than a system using a removal process (e.g. cut and cart).

TABLE 28 – NUTRIENT CONTENT AND ANTICIPATED DRY MATTER YIELD OF VARIOUS CROPS

Crop	DM Nutrient Content (%)			Normal Yield Range (DM t/ha)	Normal Nutrient Removal Range (kg/ha)		
	N	P	K		N	P	K
Dry Land Pasture (cut)	2.0	0.3	1.5	1 - 4	20 - 80	3 - 12	15 - 60
Irrigated Pasture (cut)	2.0	0.3	1.5	8 - 20	160 - 400	24 - 60	120 - 300
Lucerne Hay (cut)	3.1	0.3	2.5	5 - 15	155 - 465	15 - 45	125 - 375
Maize Silage	2.2	0.5	2.0	10 - 25	220 - 550	50 - 125	200 - 500
Forage Sorghum	2.2	0.3	2.4	10 - 20	220 - 440	30 - 60	240 - 480
Winter Cereal Hay	2.0	0.3	1.6	10 - 20	200 - 400	30 - 60	160 - 320
Seed Barley	1.9	0.3	0.4	2 – 5	38 – 95	6 – 15	8 – 20
Seed Wheat	1.9	0.4	0.5	2 – 5	38 - 95	8 - 20	10 – 25
Triticale	1.9	0.4	0.6	1.5 - 3	29 - 57	6 - 12	9 – 18
Rice	1.4	0.3	0.4	4 - 8	56 - 112	12 - 24	16 - 32
Seed Oats	1.5	0.3	0.4	1 - 5	15 - 75	3 - 15	4 – 20
Grain Sorghum	2.0	0.3	0.3	2 - 8	40 - 160	6 - 24	6 – 24
Grain Maize	2.0	0.3	0.4	2 - 8	40 - 160	6 - 24	8 – 32
Chickpea	4.0	0.4	0.4	0.5 - 2	20 - 80	2 - 8	2 – 8
Cowpea	3.0	0.4	2.0	0.5 - 2	15 - 60	2 - 8	10 – 40
Faba Bean	4.0	0.4	1.2	1 - 3	40 - 120	4 - 12	12 – 36
Lupins	4.5	0.3	0.8	0.5 - 2	22.5 - 90	1.5 - 6	4 – 16
Navy Bean	4.0	0.6	1.2	0.5 - 2	20 - 80	3 - 12	6 – 24
Pigeon Peas	2.6	0.3	0.9	0.5 - 2	13 - 52	1.5 - 6	4.5 – 18
Cotton	2.0	0.4	0.8	2 - 5	40 - 100	8 - 20	16 – 40
Asparagus	0.4	0.4	2.5	0.5 - 2	2 - 8	2 - 8	12.5 - 50
Beans	3.1	0.3	2.6	4 - 8	124 - 248	12 - 24	104 - 208
Beetroot	4.2	0.3	4.0	5 - 15	210 - 630	15 - 45	200 - 600
Broccoli	3.9	0.5	3.0	5 - 15	195 - 585	25 - 75	150 - 450
Cabbage	3.5	0.4	4.0	5 - 15	175 - 525	20 - 60	200 - 600
Carrot	0.9	0.4	1.7	5 - 15	45 - 135	20 - 60	85 - 255
Cauliflower	3.6	0.5	4.3	5 - 15	180 - 540	25 - 75	215 - 645
Celery	2.1	0.3	4.0	5 - 15	105 - 315	15 - 45	200 - 600
Lettuce	4.0	0.5	6.0	5 - 15	200 - 600	25 - 75	300 - 900
Onion	1.3	0.4	2.2	5 - 15	65 - 195	20 - 60	110 - 330
Peas	2.0	0.2	1.2	4 - 8	80 - 160	8 - 16	48 - 96
Potato	2.5	0.2	2.2	5 - 15	125 - 375	10 - 30	110 - 330
Tomato	3.6	0.7	4.7	5 - 15	180 - 540	35 - 105	235 - 705

Sources: Reuter, D.J., Robinson, J.B. (eds) (1997) and National Research Council (1984.

7.4. Calculating Sustainable Application Rates

It is good agronomic practice to know the nutrient status of reuse areas. It is good environmental practice to know both the application rates and the nutrient removal and storage rates through crop harvest, phosphorus storage, nitrogen volatilisation and other acceptable losses. This information needs to be known in order to manage a reuse area in an environmentally sustainable manner.

The first step in the risk assessment process is nutrient quantification for the reuse area. The recommended method for estimating nutrients and salts from piggeries and feedlots is mass balance considering inputs and outputs. This estimates the net mass of nitrogen, phosphorus and salt added to reuse areas as the difference between additions via effluent, solid by-product and/or inorganic fertiliser applications and removal via crop harvest and acceptable losses (nitrogen volatilisation and salt leaching).

The mass balance could be a desktop study (e.g. PigBal, BeefBal, MEDLI) or could use physical measurements coupled with a desk top study (e.g. the quantity of nutrients applied could be determined by effluent or solids analysis and measurement of application rates. This would then be compared with the expected nutrient removal rate by cropping). In the absence of site-specific mass balance modelling, figures given in Sections 6.3.12 and 6.6.9 can be used.

In the simplest form, a system is sustainable if nutrient removal by crop harvest matches nutrient applications, the soil resource is maintained or improved, and the environment and public health is protected. However, there are good arguments for modifying this definition for the reuse of effluent or solid by-products from piggeries and cattle feedlots. For example, most soil types have a significant capacity to store phosphorus. Since many Australian soils are also inherently deficient in phosphorus, it makes good agronomic sense to apply phosphorus to the soil at rates exceeding the nutrient uptake by cropping. Also, most Australian soils used for crop production have a good capacity to retain phosphorus. Salt tends to be more complex since growing plants remove relatively small amounts of salts.

The Mass Balance Equation for reuse areas is:

Crop Uptake + Allowable Losses + Safe Soil Storage = Amount Applied

To solve this equation, it is necessary to quantify the following parameters, considering the management practices employed and the natural resources of the site:

- Allowable losses
- Safe phosphorus storage capacity

Allowable losses may include:

- N volatilisation during and after application
- Leaching – provided it does not exceed an acceptable level OR degrade the groundwater source.

Availability of nutrients also needs consideration. Not all the nutrients in effluent and solid by-products are available immediately for plant uptake and part may not become available for several years. Thus to meet the crop requirements for nutrients and subsequently maximise uptake, the total mass of nutrients applied in organic fertilisers (manure and effluent) may need to be significantly greater than actual plant uptake. This practice is particularly relevant for the application of solid manure, where there are environmental and economic benefits in applying manure every three to four years via reduced cultivation and soil compaction. When applying organic fertilisers, the mineralisation of organic components (nitrogen and phosphorus) needs to be considered to ensure there are enough available nutrients to meet crop demand. The addition of inorganic fertiliser may also need to be considered in the years between manure application to ensure the crops requirements are met and thus nutrient uptake is maximised.

One of the advantages of organic fertilisers over many inorganic fertilisers is that they release nutrients for plant use over time and these nutrients become available to the crop as they are required, rather than all being available when the crop is first sown.

7.5. Control Measures and Practices to Minimise Export of Nutrients

Measures typically used to minimise nutrient export from reuse areas include:

- Vegetative filter strips located downslope of the reuse area.
- Terminal ponds located downslope of the reuse areas.
- Contour banks installed on sloping land and runoff diversion banks/ditches upslope of reuse area.
- Maintaining continuous ground cover.
- Direct injection of slurry and incorporation of solids as soon as possible after application.
- Use of phosphorus sorbing treatments (e.g. red mud, ferric chloride).
- Using sound reuse practices (including irrigation scheduling and the use of soil moisture sensors).

These measures are effective at both reducing soil erosion and filtering nutrients from runoff water. However control measures, such as vegetative filter strips and terminal ponds should not be used as a "quick-fix" for poor reuse practices. They provide secondary environmental protection after sustainable reuse based on mass balance principles and/or monitoring. Employing these control measures at intensive animal operations with sustainable application rates, is likely to achieve lower nutrient losses than those expected from 'conventional' cropping practices using inorganic fertilisers.

The application method also needs to match the land resources of the reuse area. For example, flood irrigation will not work effectively on uneven sloping land.

7.5.1. Vegetative Filter Strips

Redding and Phillips (2002) describes vegetated filter strips (VFS) as strips of dense grass between a reuse area and a protected area (eg a surface water resource). Research with various sources of phosphorus, including effluent phosphorus, indicates that VFS can almost eliminate phosphorus transport in runoff water by reducing runoff volume and concentration. VFS reduced the concentration of phosphorus in the runoff from piggery effluent reuse areas by at least 78% (using a 5 metre VFS based on flow weighted averages; 40 metre slope length). There was also an 80% reduction in runoff volumes (using a 2 metre VFS). This translates to reductions of around 95% where both infiltration and particle trapping processes are active, and around 80% under more extreme rainfall events.

A VFS should:

- Be used to protect surface water bodies adjacent to reuse areas.
- Be planted with runner developing, non-clump forming grass species. Kikuyu grass is ideal.

- Be relied on to immobilise phosphorus only where soil losses are less than 50-70 t/ha/yr. For higher soil loss rates, additional measures should be in place.
- Be established as close as possible to the reuse area to minimise additional runoff through the VFS.
- Be located before the convergence of runoff.

Karssies and Prosser (1999) recommend a range of VFS widths appropriate for various conditions. Where slope lengths above the VFS are greater than 200 m these designs will not be effective. The recommended widths will also be ineffective where flow concentrates in depressions before entry into the filter strip.

TABLE 29 - GRASS FILTER STRIP WIDTHS (m) FOR TYPICAL VALUES OF SOIL LOSS AND FILTER GRADIENTS (ADAPTED FROM KARSSIES AND PROSSER, 1999 BY REDDING & PHILLIPS, 2002)

Soil Loss	Filter Strip Slope								
(t/ha/yr)	1	2	3	4	5	6	7	8	9
10	5	5	8	8	9	9	10	10	10
20	6	12	15	15	15	16	16	16	16
30	12	18	21	21	22	22	22	23	23
40	18	24	27	27	28	28	29	29	29
50	25	>30	>30	>30	>30	>30	>30	>30	>30

7.5.2. Terminal Ponds

Terminal ponds located at the bottom of reuse areas catch runoff water. They are designed to catch the first flush of runoff from a paddock. The principle is to trap some dissolved and eroded nutrients that can then be re-irrigated.

It is difficult to determine what sized runoff event needs to be caught by a terminal pond. It is not possible to catch all of the runoff from large rainfall events, because the size of the pond would become extremely large. The National Guidelines for Beef Cattle Feedlots suggest that terminal ponds should catch the first 12 mm of rainfall runoff, plus the irrigation tailwater runoff. Catching 12 mm of runoff requires a storage capacity of 0.12 ML/ha of reuse area.

Terminal ponds need to have a by-wash that directs flows after the ‘first flush’ around the pond. The by-wash needs to be up-stream of the dam wall.

Any flood irrigation system or outdoor pig production facility should have a terminal pond.

7.5.3. Contour Banks and Diversion Banks

Banks constructed along height contours on sloping areas reduce runoff water velocity. They capture and redirect runoff from smaller areas of a paddock, preventing runoff from concentrating into larger streams that erode large volumes of soil. Diversion banks and ditches upslope of effluent irrigation areas will reduce the amount of runoff.

7.5.4. Maintaining Continuous Groundcover

Maintaining continuous groundcover either by having a pasture based reuse system or utilising conservation tillage practices reduces water velocity and soil movement. This reduces nutrient removal by soil erosion.

7.5.5. Direct injection of slurry and solids incorporation

Using slurry tankers with injection will not only reduce the risk of nutrient export, but also has the added advantages of minimising nitrogen loss and eliminating odour issues associated with application. Incorporating solids as soon as possible after application will also minimise the risk of nutrient export.

7.5.6. Use of phosphorus sorbing treatments

More advanced treatment options include the use of phosphorus sorbing treatments, such as red mud and ferric chloride. Red mud is a waste product from aluminium refining that can be used to precipitate out phosphorus, but it is highly alkaline. Ferric chloride is used to purify surface water. The purification occurs by flocculating suspended materia, removing organic substances, reducing phosphates and removing heavy metals via coprecipitation.

7.5.7. Effluent Irrigation Management Practices

The irrigation practices used affect nutrient export from reuse areas. For example, irrigating effluent when the soil is dry reduces the risk of runoff during or soon after application. Also, nutrient leaching can be reduced by applying effluent when crops are actively growing and can take up the nutrients and by applying smaller amounts of effluent frequently rather than in large depths occasionally. With sound irrigation scheduling, the application of effluent to soil can be matched to soil storage characteristics (soil moisture sensors will also assist with this scheduling).

7.6. Risk Assessment: Design and Management of Reuse Areas

This section provides information to decide the risk class of various reuse design and management options. These design and management options include:

- Sizing reuse areas for sustainable reuse
- Using appropriate application methods for effluent and solid by-products
- Providing safeguards for minimising nutrient exports via processes such as erosion and leaching

Sizing reuse areas and using appropriate application methods for applying effluent and solid by-products are primary methods for minimising the environmental risks associated with reuse. These design and management options, as well as the risk associated with the nutrients in effluent and solids (Section 6.7) are evaluated against the natural resources (surface water, groundwater and soil) vulnerability assessment criteria for the site in Section 11.2.

Providing design or management based safeguards for minimising nutrient exports is a secondary method for reducing nutrient exports by reducing resource vulnerability. Consequently, these safeguards are dealt with in Section 10.

7.6.1. Size of Land Area and Application Rate

Select the appropriate risk category for each sub-heading below. The **highest** risk weighting for Section 7.6.1 is then transferred into the "Size of Land Area" row of Table 30.

Knowledge of Size of Land Area

Low Risk	From farm or paddock maps, you accurately know the area (ha) of each effluent or manure reuse paddock under each management regime (e.g. soil properties, land use).
Medium Risk	You know the approximate area (ha) of each effluent or manure reuse paddock under each management regime.
High Risk	You do not know the area of the effluent or manure reuse paddocks.

Knowledge of Yields of Crops or Pastures Grown on Reuse Areas

Low Risk	For your property and soil type, you know typical yields for the pastures or crops grown on reuse areas.
Medium Risk	You know typical district yields for the pastures or crops grown on reuse areas.
High Risk	You do not know typical yields for the pastures or crops grown on reuse areas.

Knowledge of Nutrients Applied to Reuse Areas

Low Risk	You have calculated the nitrogen (kg/ha/yr) and phosphorus (kg/ha/yr) loading rates to reuse areas from estimated nutrient production.
High Risk	You have not calculated the nitrogen (kg/ha/yr) and phosphorus (kg/ha/yr) loading rates to reuse areas.

Nitrogen Mass Balance for Reuse Areas

Low Risk	You have calculated that the net mass of nitrogen (kg/ha/yr) applied as effluent and / or solid by-products is exceeded by the mass of nitrogen (kg/ha/yr) that plant harvest should remove.
Medium Risk	You have calculated that the net mass of nitrogen applied (kg/ha/yr) as effluent or solid by-products is equal to the mass of nitrogen (kg/ha/yr) that plant harvest should remove.

High Risk — The net mass of nitrogen applied to reuse areas (kg/ha/yr) exceeds the mass removed or you do not know the net mass of nitrogen applied to the reuse area.

Phosphorus Mass Balance for Reuse Areas

Low Risk — You have calculated that the mass of phosphorus (kg/ha/yr) applied as effluent and/or solid by-products is exceeded by the mass of phosphorus (kg/ha/yr) that plant harvest should remove plus phosphorus storage calculated from a site-specific phosphorus sorption test.

Medium Risk — You have calculated that the net mass of phosphorus applied (kg/ha/yr) as effluent or solid by-products is equal to the mass of phosphorus (kg/ha/yr) that plant harvest should remove plus phosphorus storage calculated from a site-specific phosphorus sorption test or from generic phosphorus sorption data for similar soil types.

High Risk — The net mass of phosphorus applied to reuse areas (kg/ha/yr) exceeds the mass removed plus storage calculated from a site-specific phosphorus sorption test or from generic phosphorus sorption data for similar soil types or you do not know the mass of phosphorus applied to the reuse area.

Select the highest risk weighting from the above categories to transfer to the "Size of land area and Application rate" row of Table 30.

7.6.2. Using Appropriate Effluent & Solid By-Product Application Methods

If you reuse effluent on-site, select the appropriate risk category for "Effluent Irrigation" based on the information presented below. If you reuse solid by-products on-site, select the appropriate risk category for "Solids Spreading" from the information presented below. If you reuse effluent and solids on the same area, select the risk weighting that is **highest** from either the "Effluent Irrigation" or "Solids Spreading" sections below (e.g. if you have a rating of *low* for effluent irrigation and a rating of *medium* for solids spreading, the overall risk weighting you choose for the area is *medium*).

The results then need to be transferred into the "Using Appropriate Effluent & Solid By-Product Application Methods" row of Table 30 and converted into a risk weighting. A separate copy of Table 30 needs to be developed for separate reuse areas (e.g. effluent areas V solid areas) or reuse areas posing different risks (e.g. one effluent reuse area might be low risk, another high risk).

Effluent Irrigation

Low Risk — You use a low-pressure, travelling spray or drip irrigation system or a low-pressure solid set spray or drip irrigation system or a well designed and maintained flood irrigation system that is not on sandy to sandy loam soil. The system also applies effluent evenly and at target rates.

High Risk	You use a hand-shift sprinkler or hose or a poorly designed or managed flood irrigation system (e.g. land has not been levelled or effluent is unshandied or surface soil is sandy to sandy loam).

Solids Spreading

Low Risk	The spreading method used disperses solids evenly and at target rates.
Medium Risk	The spreading method used disperses solids fairly evenly and within 20% of target rates.
High Risk	The spreading method used disperses solids unevenly or at uncontrolled rates (not within 20% of target rates).

Select the highest risk weighting from the above categories to transfer to the "Application methods" row of Table 30.

Table 30 is a template for summarising the design and management risk weightings for each design and management criterion. To complete the table, insert a risk weighting of 1, 2 or 3 against each criterion. A low risk attracts a risk weighting of "1", medium risk attracts a risk weighting of "2" and high risk attracts a risk weighting of "3". These numbers are transferred to Table 54, Table 55 and Table 56 in Section 11.2.

TABLE 30 – DESIGN AND MANAGEMENT REUSE AREA RISK ASSESSMENT SUMMARY

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)
Size of land area and Application Rate	e.g. 3
Application methods	

Transfer these values to the Risk Assessment Matrices for soils, surface water and groundwater (Table 54, Table 55 and Table 56).

8. SUSTAINABILITY INDICATORS

This section provides the technical background needed to identify appropriate indicators of environmental sustainability for effluent and solid by-product reuse. These indicators are based on current knowledge and available information. For some indicators, the knowledge of processes involving the indicator is limited, or data available regarding the relationship between the indicator and particular environmental effects is poorly defined or limited to particular geographical locations. In instances where knowledge or data is limited, regulatory authorities often apply the precautionary principle to establish a conservative guideline.

Walker and Reuter (1996) list a number of criteria for the selection of indicators to determine the health of a catchment. These criteria can be used to determine appropriate sustainability indicators of effluent and solids reuse for piggeries and cattle feedlots. They include:

- Able to be measured easily and economically
- Fewer rather than many indicators
- Able to be measured at achievable and appropriate levels of precision
- Simply quantified
- Interpretable (able to be linked directly to real questions).
- Able to indicate spatial and temporal variation.
- Able to suit all levels of enterprises.
- Easily captured.
- Total cost /ha/test.
- Existence of a standard method of estimation.
- Interpretation of criteria available – expected and threshold values.
- Significant on a catchment scale to estimate condition – for a piggery or cattle feedlot this can be ascertained on a property scale.
- Low error associated with measurement.
- Known response to land management or disturbances.
- Trend indicators are mappable – as in property management plans.
- Generic rather than diagnostic..

The appropriateness of the following indicators has been determined using an evaluation with the above criteria.

Due to the uncertainty inherent in some of these indicators, it is important that the selection and use of sustainability indicators is transparent and open to input and regular review. This is important to allow for the incorporation of emerging scientific facts. This review process should include the frequency of review, process of review (how intensively) and who reviews these (including review by industry representatives).

Much of the detail relating to this section is included in appendices (testing standards, accuracy of tests, sample handling and costs) at the back of the report. It includes a detailed study of nitrogen, phosphorus, salinity and sodicity, and a philosophical discussion on the sustainability of each of these parameters. Consequently it also provides an introduction to subsequent chapters.

The following sustainability indicators have been judged to provide the best practical and objective measures of sustainability. It is expected that in most cases they will provide a good tool for the assessment of sustainability. However, it is important to recognise that non-compliance with the standards set-down in these indicators does not necessarily indicate that a system is unsustainable. In these instances, operators of piggeries and cattle feedlots may use other indicators to demonstrate sustainability.

8.1. Nitrogen

This section is a detailed study on nitrogen and its effect on soil, including:

- Pathways for nitrogen export, including leaching and volatilisation losses.
- Forms of nitrogen (organic, ammonia, NOx) and typical ranges in effluent.
- Soil chemistry (nitrogen cycle, availability) and nitrogen dynamics in soil.
- Information on latest Australian research studies (Australian Pork Limited and the Cattle & Beef CRC funded research).
- Measuring nitrogen sustainability.
- Analysis methods (standards, costs, accuracy, sample storage & handling).

8.1.1. Introduction

Nitrogen is an essential element for plant growth. Aside from legumes and a few other species most plants source nitrogen from the soil or plant litter, or nitrogen-containing soil amendments (e.g. piggery and cattle feedlot manure and effluent).

A plant's nitrogen supply needs to be managed to prevent deficiencies during periods of high plant demand and to prevent excesses of mineral nitrogen (predominantly ammonium and nitrate) during cropping or during periods without crop. The latter is necessary to minimise the risk of nitrogen leaching to groundwater. Because plant nitrogen requirements and the release of plant-available nitrogen from soil organic reserves vary with seasonal climatic conditions, temporary nitrogen deficiencies occur in most cropping systems (Strong and Mason, 1999). However, organic fertilisers, such as feedlot and piggery manures and effluents act as a 'slow-release' fertiliser-N for plants, which can be environmentally advantageous over inorganic fertilisers that have the majority of the applied N in a form readily available to the plant soon after application.

8.1.2. Loss Pathways and Impacts of Nitrogen

Nitrogen is not conservative and is lost as gaseous ammonia (volatilisation) and as oxides of nitrogen (denitrification), following mineralisation from its organic form. Thus, calculating a nitrogen balance for piggeries and cattle feedlots from excretion through to application and reuse is a difficult process involving a number of assumptions in estimating losses throughout the process. These loss pathways are discussed below.

Volatilisation from feedlot pads and piggery sheds

A significant amount of the nitrogen in pig and cattle manure is in the ammonium and urea forms and is readily lost by ammonia volatilisation soon after excretion. Details of the nitrogen loss from piggery sheds are covered in detail in Section 6.3.7 and for feedlots in Section 6.6.5.

Volatilisation losses from treatment ponds

Nitrogen losses by ammonia volatilisation from ponds, depends mainly on the ammonium concentration at the pond surface, surface area of the pond and pond chemistry (e.g. pH). These, and other factors determine the transfer at the liquid/air interface. For more information on nitrogen loss from effluent ponds, see Section 6.3.7.

Volatilisation Losses During Irrigation

Most nitrogen losses during irrigation are due to ammonia volatilisation. The type of irrigation system affects the volatilisation rate. An irrigation system producing small droplets is likely to produce higher volatilisation rates, because of the greater total surface area of the droplets.

Nitrogen losses during irrigation also vary with pH. Henderson *et al.* (1955) showed that at a neutral pH (piggery effluent) nitrogen losses ranged from about eight to ten percent.

Smith *et al.* (2001) reported ammonia losses from a range of overseas research. These losses ranged from 14 - 38% for piggery effluent reuse. The research by Smith *et al.* (2001) using piggery effluent on a winter and summer crop rotation in south-eastern Australia showed that about 12% of the total nitrogen was lost by ammonia volatilisation and represented a less significant loss pathway than previously thought.

This research studied a centre-pivot irrigator applying 18 mm of effluent every three days. The irrigator operated 24 hours a day. When these losses were split into daytime and night-time losses, they corresponded to 21% and 3% respectively. Night-time effluent irrigation is not regularly practiced for intensive animal operations in Australia, being discouraged from an odour dispersion perspective. Smith *et al*. (2001) also states that the average 12% loss ignores losses from the boom, which account for approximately 7% loss.

More information on nitrogen losses during irrigation can be found in Section 7.1.

Denitrification Losses of Nitrogen

Nitrogen can be lost via denitrification (the conversion of nitrate-N into gaseous nitrogen forms of N_2O, and N_2 that are lost to the atmosphere). Denitrification occurs if the soil becomes anoxic (i.e. oxygen deficient), usually by water logging. Also, readily available carbon must be present to fuel the bacteria responsible for the denitrification conversions.

Smith *et al.* (2001) reported that denitrification losses from agricultural systems irrigated with ammonium based fertilisers are typically 20 – 40%, but losses as high as 70% have been reported. They reported that no quantitative estimates of nitrogen losses by denitrification are available for soil-plant systems irrigated with piggery effluent under Australian weather conditions. This study showed that biological denitrification could not remove excess nitrogen irrigated in piggery effluent.

Nitrate Leaching

A key indicator of groundwater contamination is elevated nitrate-nitrogen levels, which suggest leaching of surplus nitrogen from the soil to the groundwater.

Rainfall and irrigation move nitrogen-nitrates deeper into the soil profile (leaching). Leaching is more likely in soils with a rapid internal drainage, such as sands and loams. In vertosols, such as those found on the Darling Downs in Queensland, high nitrate levels are evident in sub-soil layers due to successive recharge of soil water during periods of summer fallow, combined with incomplete crop use of sub-soil nitrate (Waring and Teakle, 1960). Other factors that increase nitrate leaching include failures of the plant roots to penetrate the sub-soil and preservation of macropores for increased and preferential flow of soil nitrate as observed in zero-till systems (Dalal, 1992; cited by Strong and Mason, 1999).

Over-loading reuse areas with nitrogen may contribute to nitrate-nitrogen pollution of groundwater. Smith *et al.* (2001) reported that applying nitrogen in piggery effluent at rates exceeding crop requirements substantially increases nitrate leaching beyond the root zone. This work involves applying effluent at a rate meeting the crop water requirements during the summer growing period. This equated to a nitrogen application exceeding 2500 kg N/ha/yr. This proved that piggery effluent should not be irrigated at rates meeting crop water requirements without dilution with clean water.

Note: The root zone depth depends on the crop type, soil, climatic condition and whether the crop is irrigated. The depth of the root zone may sometimes be 1.5 – 2.0 m and even further (e.g. dryland lucerne). Thus, sampling below the root zone may not always be feasible.

Nitrogen Export in Runoff

Nitrogen in the nitrate and ammonium forms is highly soluble and readily dissolves in rainfall runoff. It can also be exported in runoff as suspended organic and inorganic material. Applying effluent at rates exceeding the soil's infiltration capacity may cause direct runoff from irrigation areas. The application of effluent and solid by-products immediately prior to rainfall increases the potential for soluble nitrogen export. Provided appropriate soil erosion control measures are in place, incorporating solid by-products immediately after application reduces the risk of nitrogen and other nutrients and salt losses to surface waters.

The Adverse Impacts of Nitrogen Loss

Nitrate levels exceeding 10 mg/L of NO_3-N (or 45 mg/L NO_3) in water for human consumption includes methaemoglobinaemia, a human health problem in infants (blue baby syndrome). This condition reduces the oxygen transport capacity of the blood. Similar toxic effects have been observed in animals drinking high nitrate waters. Pigs are generally more sensitive to high nitrate waters than cattle and sheep.

Elevated nitrate levels in surface waters (>3 mg/L) with elevated phosphorus levels, may also lead to eutrophication. The result of this is excessive algal growth, depleted oxygen levels and the possible death of fish and other aquatic organisms.

8.1.3. Forms of Nitrogen

Organic nitrogen is either ingested diet material, excreted undigested plant material or soil organic matter. Inorganic nitrogen includes ammonium (NH_4^+), nitrate (NO_3^-), the gaseous form of ammonia (NH_3), nitrous oxides (N_2O, NO) and dinitrogen (N_2). The nitrogen forms available for plant uptake are NH_4^+ and NO_3^-, whilst all the other forms reflect transformation or loss processes (Gardner *et al.* 1994). Total oxidized nitrogen is the sum of the nitrate and nitrite nitrogen.

Organic nitrogen is organically bound in the tri-negative oxidation state. It includes such natural materials as proteins and peptides, nucleic acids and urea, and numerous synthetic organic materials. It excludes all organic nitrogen compounds. Organic nitrogen and ammonia nitrogen can be determined together and are known as the “Kjeldahl nitrogen”, a term reflecting the determination technique.

The organic nitrogen forms are contained primarily in the faeces. They are either readily available for conversion to the inorganic NH_4^+ form by soil micro-organisms (the mineralisation process) or slowly available to the mineralisation process. In the faeces, undigested proteins and unabsorbed sugars are readily available for decomposition whilst structural carbohydrates in plant material (e.g. hemicelluloses) are slowly available. Plant cell wall material such as lignin is highly resistant to microbial decomposition (Gardner *et al.,* 1994).

The other (and major) pathway of nitrogen excretion is via the urine. About 70% of nitrogen in urine is present as urea and about 30% is present as readily mineralised organic compounds (e.g. proteins, amino acids etc). Urea is readily broken down under exposure to moisture, air, and the urease enzyme, producing aqueous ammonium (NH_4^+) (Gardner *et al.* 1994).

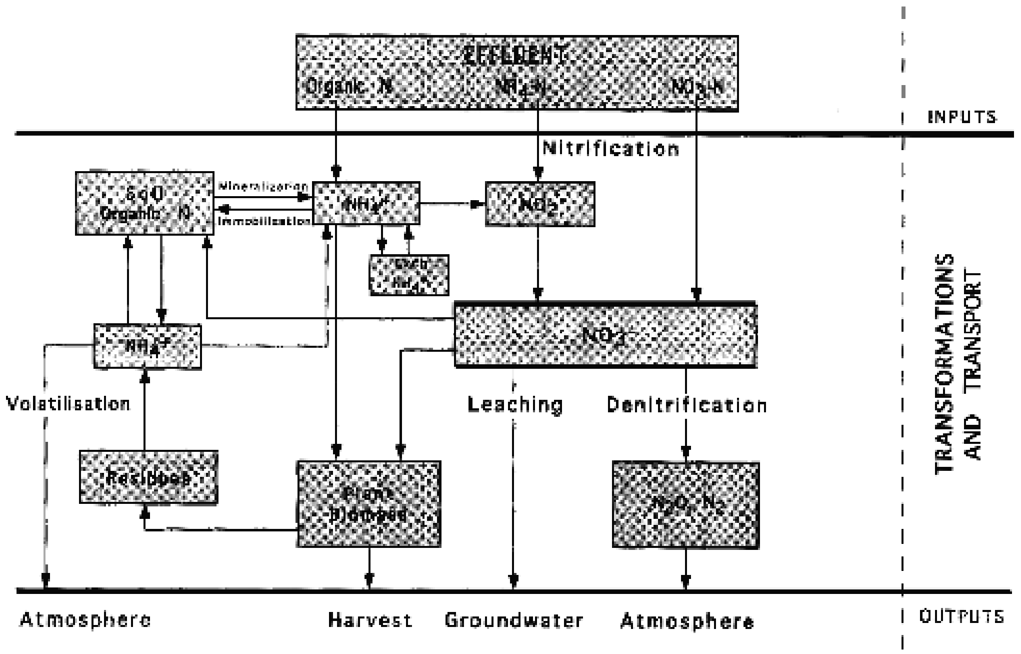
8.1.4. Soil Nitrogen Dynamics

Nitrogen applied to the soil as effluent and manure can undergo a number of transformations including:

- Mineralisation – the decomposition of organic nitrogen to ammonium (NH_4^+).
- Immobilisation of inorganic forms of nitrogen by plants and microorganisms to form organic nitrogen compounds
- Nitrification – the oxidation of ammonium (NH_4^+) to nitrite (NO_2) and then into nitrate (NO_3).
- Denitrification of nitrate (NO_3) to nitrous oxide and nitrogen gas.
- Hydrolysis of urea into the ammonium (NH_4^+) form.

Figure 2 shows the nitrogen pathways of effluent (or manure) on reuse areas. Each of these pathways is discussed in further detail below.

FIGURE 2 – NITROGEN PATHWAYS OF EFFLUENT IN LAND REUSE AREAS (GARDNER *ET AL.* 1994).



Mineralisation and Immobilisation

The amount of nitrogen mineralised or immobilised depends on the organic matter forms, temperature and soil moisture. Under ideal conditions a rapid increase in microbial population provides a large sink for nitrogen for use in cell synthesis (Kruger *et al.,* 1995).

Skerman (2000) provides an example mineralisation decay series of Pratt *et al.* (1973) (cited by Casey and Gardner (1995)). For a constant annual manure application rate, the proportion of the annual nitrogen application that is mineralised increases from 35% in year 1, to 55% in year 5, to 65% in year 10, to 79% in year 20.

Skerman (2000) identifies that constant annual manure applications based on balancing nitrogen provide insufficient mineralised plant available nitrogen during the early years. More closely balancing nitrogen availability to crop requirements involves applying higher rates of manure during the early years and gradually decreasing annual manure applications. However Skerman (2000) cautions against an application strategy based solely on balancing crop nitrogen requirements since this may overload the soil-crop system with phosphorus or salts.

Due to the slow release of nitrogen via the mineralisation process, the one-off application of solid manure every few years, with inorganic fertiliser ‘top-ups’ as required between years as required may be a more sustainable process, as it will assist in maintaining groundcover (no working in of the manure required) and reduced soil compaction.

The microbial biomass can use ammonium nitrogen provided there is a carbon source in the effluent or manure. This process is termed immobilisation – the opposite of mineralisation and the balance between the two processes is largely determined by the C:N ratio of the added material (Gardner *et al.* 1994). As a general rule of thumb, if the C:N ratio exceeds 25, there is a net immobilisation because the carbon stimulates microbial growth incorporating all of the nitrogen added by the effluent or manure into the microbial cell structure (Van Veen *et al.* 1984; White, 1987 – cited by Gardner *et al.,* 1994).

Nitrification and Denitrification

Nitrification is an aerobic process, which transforms relatively immobile ammonium into nitrate. Temperature and oxygen supply govern the nitrification rate. Under aerobic, warm conditions there is almost complete conversion of ammonium to nitrate in the surface soil within a few days of effluent application (Kruger *et al.,* 1995).

As explained earlier, denitrification occurs under anaerobic conditions. As free oxygen is in short supply, bacteria use nitrite and nitrate as a source of oxygen and produce nitrous oxide and nitrogen that are lost from the soil as gases.

Part of the NH_4^+ retained in the soil solution or the exchange sites is transformed via bacteria into the nitrate NO_3^- form (nitrification). NO_3^- is the most mobile form of nitrogen in soils because of its negative charge. NO_3^- is available for uptake by plant roots; for leaching below the crop root zone, or for transformation by soil bacteria into the gaseous N_20 or N_2 forms (the denitrification process) that are then lost to the atmosphere (Gardner *et al.,* 1994).

Both nitrification ($NH_4^+ \rightarrow NO_3^-$) and denitrification ($NO_3^- \rightarrow N_3O$, N_2) are strongly influenced by temperature, pH and soil water content (Haynes and Williams 1993). Denitrification only occurs if the soil becomes anoxic (i.e. oxygen deficient), usually by water logging. Also, readily available carbon must be present to fuel the bacteria that convert the nitrogen (Gardner *et al.* 1994).

Urea Transformations

Urea transformations and the partitioning between NH_4^+ and NH_3 forms, loss of NH_3 as gas, and retention of NH_4^+ by soils depend largely on pH, temperature and cation exchange capacity (CEC). With increasing soil temperature and alkalinity, transformation of aqueous ammonium into NH_3 increases (Freney *et al.,* 1983). Since urea hydrolysis always involves increasing pH due to the release of hydroxyl ions, poorly buffered soils (e.g. sandy textures with low organic matter) with low CEC lose the most urea as NH_3 gas (on a percentage basis) (Whitehead and Raistrick, 1993). This is true irrespective of whether the urea is sourced from animal effluent or from synthetic fertilisers (Gardner *et al.,* 1994).

It is important to realise that the organic material decomposition is driven by the carbon demand of soil microflora for energy and as a building block for new cell growth. Nitrogen releases as ammonium and phosphorus and other inorganic material from the organic material is only incidental to the microbial growth process (Gardner *et al.,* 1994).

8.1.5. Review of Recent Studies on the Reuse of N in Effluent and Manure

Smith *et al.* (2001) investigated nitrogen losses during irrigation of piggery effluent. The effluent was irrigated onto a rotated cropping regime of oats and maize in south-eastern Australia. Effluent irrigation occurred approximately every three days during the summer cropping period. Effluent was applied at a depth of approximately 18 mm.

Ammonia losses were approximately 12% of the total nitrogen load applied. Applying 'high strength' piggery effluent at rates meeting the crop's water demand applied nitrogen at levels exceeding the crop's nitrogen requirement. Volatilisation and denitrification were insufficient to remove the excess nitrogen. This demonstrated that effluent nitrogen applications at rates exceeding crop requirements substantially increases nitrate leaching beyond the root zone.

This work used a centre-pivot irrigator operating 24 hours a day. When losses were split into daytime and night-time losses, they corresponded to 21% and 3% respectively. The 24-hour a day application of effluent for intensive animal operations is not regularly used in Australia and is discouraged from an odour dispersion and impact perspective. Smith *et al*. (2001) also states that average 12% loss ignores losses from the boom, which account for approximately 7% loss.

Smith and Snow (2001) also studied the effectiveness of an overland flow system at removing the nitrogen. They found that at least 48% of the nitrogen applied was lost either as volatilisation or denitrification. They also concluded that the overland flow process did not treat piggery effluent to a sufficient quality for direct discharge into the environment.

Work by the Cattle and Beef CRC (Klepper *et al*., 2001) reported "Most nutrients in feedlot manure remain unavailable due to complexation and the form in which they are present, until the manure is mineralised and nutrients are released in the organic form. The low recovery of nitrogen, phosphorus and sulphur derived from manure is due to the slow mineralisation rate with nitrogen being the most limiting factor." Since phosphorus is added at a disproportionately greater rate than nitrogen (plants generally require a N:P ratio of 5:1 or greater, whereas manure has a ratio of 2:1) their experiments showed that nitrate-N levels were exhausted after two years of trials, whereas phosphorus levels were seven times the initial concentrations. "Balancing the nutrition of a manure fertilised, high yielding fodder cropping system (with the addition of nitrogen) is the best way of managing the nutrient imbalance contained in feedlot manure and effluent. With intensive double-cropping this work showed that up to 600 kg/ha/yr of nitrogen was removed in plant yield. With successive forage crops and minimum cultivation, nutrient uptake is maximised, soil aggregation is maintained and losses in surface runoff and deep percolation are minimised.

8.1.6. Analysis Methods (Including Accuracy)

Total nitrogen and 'mineral nitrogen' are of particular interest from a land use and soil fertility viewpoint. Nitrate, and to a lesser extent ammonium, are important sources of nitrogen for plant growth, while total nitrogen provides a measure of the quantity of nitrogen that can be 'mineralised' under appropriate conditions (Rayment and Higginson 1992).

Measurement of total N, based on wet oxidation (Kjeldahl, 1883) has found wide acceptance (Bremner and Mulvaney, 1982). Total Kjeldahl nitrogen is the sum of the ammonia nitrogen and organic nitrogen in the sample. It does not include nitrogen in the form of azide, azine, azo, hydrazone, nitrate, nitrite, nitrile, nitro, nitroso, oxime, and semi-carbazone. There are two Kjeldahl methods - macro and semi-micro.

Mineral nitrogen measures ammonium-N and nitrate-N in soils. However, care is needed in the sampling and determination, since rapid transformations can alter their apparent concentrations.

Appendix A has more details on nitrogen analysis methods.

8.1.7. Measuring Nitrogen Sustainability

Nitrogen sustainability in reuse areas can be measured by the amount of soluble and very mobile nitrate-nitrogen below the plant active root zone. Once nutrients go below the plant root zone, they can no longer be utilised by the plant. Unacceptable nitrate-nitrogen leakage can be measured either by setting a limiting soil solution based concentration at the base of the root zone, or by comparing base of root zone nitrate-nitrogen levels for reuse areas with those of similar soils not receiving effluent or solid by-products.

In most cases, subsoil nitrate-nitrogen concentrations exceeding a soil solution concentration of 10 mg nitrate-N/L are likely to produce some nitrogen leaching losses. The 10 mg/L nitrate-N is based on the *Australian Water Quality Guidelines for Fresh and Marine Waters* (ANZECC, 1992) which state that nitrate-nitrogen concentrations should not exceed the 10 mg/L level in groundwater used for human consumption. It is intended that future water quality guidelines will be largely based on recommendations from the World Health Organization (WHO). Until these guidelines are revised and endorsed, users should apply the guidelines from the *Australian Water Quality Guidelines for Fresh and Marine Waters* (ANZECC, 1992).

The *Australian Drinking Water Guidelines* (NHMRC and ARMCANZ, 1996) provide an authoritative Australian reference on good water drinking quality and include a wide range of characteristics for drinking water quality. They are not intended as guidelines for environmental water quality, nor as the document stresses, should they ever be seen as a licence to degrade the quality of a drinking water supply to a guideline value.

Applying a drinking water quality standard is likely to be overly stringent in many cases since the groundwater under reuse sites is unlikely to be used for human drinking water and it assumes there is no further losses or dilution before it reaches the groundwater. This limit is commonly exceeded in normal agricultural soils. Vertosols, for example, can have relatively high nitrate-nitrogen levels in their natural state. When assessing the sustainability of a reuse practice in terms of nitrogen levels, a number of factors need consideration, including the value or use of surrounding groundwater resources (human consumption, animal consumption, irrigation etc), the depth to groundwater, soil type overlaying the groundwater (e.g. clay) and baseline levels of nitrate-nitrogen in soil below the active root zone.

Consequently, a nitrate-nitrogen limit of 10 mg/L below the active root zone is suggested only as a trigger for further investigation. This further investigation would involve the comparison of monitoring results from the reuse area with those of the same soil that has not had effluent or manure applied (e.g. under a fenceline). If the level of nitrate below the active root zone shows signs of build-up over-time (nitrate bulges), the reuse practices employed will need review in line with the forward management plan of the operation. Thus comparing nitrate-nitrogen monitoring results against baseline data provides a measure of the nitrogen sustainability of a reuse area.

Other matters to consider when determining the sustainability of the reuse practice in terms of nitrogen include the risk of nitrate moving off-site in surface water and groundwater, the quality (value) of the groundwater and the amount of deep drainage of the soil of the reuse area. These need to be evaluated as part of the risk assessment of the reuse area.

The amount of deep drainage will vary with soil type, rainfall, the amount of effluent or fresh water irrigated and the type of crop production. For example, an improved pasture, with a total of 750 mm of rainfall and effluent irrigation, deep drainage is likely to be 10mm/yr for a black vertosol and 150 mm/yr for a loamy-sand. With 10 mg/L of nitrate-N in the deep drainage, this represents a loss of 1 kg of N/ha/yr and 15 kg of N/ha/yr for the black vertosol and the loamy sand respectively.

The depth of the root zone depends on the crop type, soil depth, climatic condition and whether the crop is irrigated. In some cases the active root zone depth may be 1.5 – 2.0 m and even deeper (e.g. dryland lucerne). Thus, sampling below the root zone may not always be practically and economically feasible. Sampling to a depth of at least 60 cm is recommended, although deeper sampling (to the base of the root zone) may be required if there are concerns about nitrate leaching.

For different soil types Skerman (2000) calculated nitrate-nitrogen concentrations equivalent to 10 mg/L of nitrate-N in soil solution (Table 31). It should be noted that soil nitrate-nitrogen concentration levels from effluent and manure reuse areas and indeed from conventional cropping systems using inorganic fertiliser, will often exceed those shown in Table 31.

Soil nitrate-N (mg/kg) = Soil gravimetric moisture conc. at field cap. (g water/g soil) x Soil sol. nitrate-N (mg/L)

TABLE 31 – NITRATE-NITROGEN CONCENTRATIONS CORRESPONDING TO A SOIL SOLUTION NITRATE-NITROGEN CONCENTRATION OF 10 mg/L AT FIELD CAPACITY.

Soil Texture	Soil Gravimetric Moisture Content at Field Capacity (g water / g soil)	Limiting Soil Nitrate-Nitrogen Concentration (mg NO_3-N / kg soil)
Sand	0.12	1.2
Sandy-loam	0.15	1.5
Loam	0.17	1.7
Clay-loam	0.20	2.0
Light Clay	0.25	2.5
Medium Clay	0.35	3.5
Self-Mulching Clay	0.45	4.5

Monitoring nitrate-nitrogen levels throughout the soil profile provides an excellent indication of sustainability. Once nitrate-nitrogen has moved below the plant root zone, it is no longer available for plant uptake, but can leach to groundwaters. A limit of 10 mg/L nitrate-nitrogen at the base of the root zone is suggested simply as a trigger for further investigation. This further investigation would include a comparison of analysis results from the base of the root zone base for reuse areas compared to areas with the same soil type that have not received effluent or manure (e.g. under fenceline). The 10 mg/L offers the highest level of protection for maintaining a groundwater resource to a human drinking water standard.

The total nitrogen concentration of the surface soil measures the amount of nitrogen potentially available for plant uptake. Surplus nitrogen may trigger a drop in soil pH, which has implications for nutrient availability. High nitrogen concentrations may also trigger excessive vegetative growth and delayed crop maturity. Mineralisable nitrogen is another measure of the quantity of nitrogen that will become available in the future.

8.1.8. Link Between Nitrogen and Organic Carbon

The quantity of total nitrogen that is available for plant uptake is strongly influenced by the organic carbon concentration of the soil, which affects nitrogen mineralisation rates. Nitrogen mineralisation is the process that transforms soil organic nitrogen into the mineral forms of nitrogen (from NH^+_4 to NO^-_2 and then to NO^-_3) that are readily available for plant uptake and leaching. Nitrogen immobilisation is the reverse process. Denitrification is the process of converting soil nitrate into the gaseous nitrogen forms (N_2 and N_2O) that may be lost to the atmosphere.

Soil carbon levels, and the dynamics between carbon and nitrogen, are extremely complex. The carbon content of the soil varies with soil depth, management, climate, soil mineral composition, soil biota, topography and the frequency of various events (e.g. fire, flood, erosion). It also depends on the rates of organic carbon addition and loss. In farming systems, continual alteration of management and cropping practices create a system where soil organic carbon levels vary (Baldock and Skjemstad, 1999).

The carbon to nitrogen ratio (C/N ratio) of the soil strongly influences the net nitrogen mineralisation rate (i.e. nitrogen mineralisation less nitrogen immobilisation) since soil microorganisms keenly compete for available nitrogen. The soil C/N ratio is typically 10-12 for an agricultural soil, but can range from 5 to 15. However, it is very constant for a given soil and is generally quite constant for similarly managed soils in a given climatic region (Charman and Roper, 2000). Organic carbon values for Victorian land under low and high rainfalls are tabulated in Table 32.

TABLE 32 – GENERAL SOIL ORGANIC CARBON CONTENTS OF SOIL CONSIDERED TO BE LOW, NORMAL AND HIGH FOR SOILS USED FOR CROP AND PASTURE PRODUCTION IN AREAS OF LOW AND HIGH RAINFALL IN VICTORIA

Soil Organic Carbon Status	Low Rainfall		High Rainfall	
	Crops	Pasture	Crops	Pasture
Low	<9	<17.4	<14.5	<29.0
Normal	9-14.5	17.4-26.2	14.5-29.0	29.0-58.1
High	>14.5	>26.2	>29.0	>58.1

Source: Baldock and Skjemstad (1999).

The overall effect of a high C/N ratio is a net immobilisation of soil nitrogen, which reduces the amount of nitrogen available for plant uptake. This occurs because soil microorganisms quickly take-up readily available nitrogen. Ultimately the amount of readily oxidisable carbon becomes limiting to microbial growth. Nitrates are released, making nitrogen available for uptake by plants grown on the area. Adding a crop residue with a lower C/N ratio shortens the length of time that immobilisation exceeds mineralisation, allowing for more rapid recycling of the nitrogen from these residues (Strong and Mason, 1999).

A high C/N ratio indicates that the addition of nitrogen or green manure crops will be beneficial. A low C/N ratio indicates that adding mature plant material e.g. cereal stubble will not be detrimental (Charman and Roper, 2000). The C/N ratio of plant residues ranges from approximately 20-30:1 for legumes and farm manure, to 100:1 in straw residues to 400:1 in sawdust (Brady, 1984).

8.1.9. Organic Carbon as a Measure of Sustainability

Although the C/N ratio provides a guide to the future availability of nitrogen, it does not provide the direct indication of sustainability provided by nitrate-nitrogen levels. However, it is worth monitoring organic carbon levels in the topsoil because of the important role organic carbon plays in maintaining the structural stability of the soil (Baldock and Skjemstad, 1999).

Total nitrogen, mineralisable nitrogen and C/N ratio all provide a guide to the quantity of nitrogen that could become available in the future. However, they are not good indicators of sustainability since they do not necessarily identify when a large amount of nitrogen is available for plant uptake and for leaching.

Soil organic carbon levels in the topsoils of reuse areas should show a stable or increasing trend over time.

Soil organic carbon is usually measured using a wet or dry oxidation procedure. In the wet process, soil organic carbon is converted to carbon dioxide using an oxidant (usually dichromate). The amount of oxidant used indicates the carbon content of the sample. Providing the samples are heated during the analysis, oxidation is considered to be complete. In the dry process, samples are heated in a stream of oxygen, which converts all carbon to carbon dioxide. The carbon content of the sample is then determined. If the soil samples contain carbonate, correction for inorganic carbon is needed to find organic carbon. This can be determined by finding total carbon and inorganic carbon on paired samples and subtracting the difference or by removing inorganic carbon with acid pre-treatment before determining the carbon content of the sample (Baldock and Skjemstad, 1999). Kalembasa & Jenkinson (1973) (cited by Baldock and Skjemstad, 1999) showed that dry oxidation methods were more precise than wet oxidation methods. Baldock and Skjemstad (1999) state that a dry combustion automated analyser measuring carbon dioxide using an infrared detector is the best method, providing accurate estimates of inorganic carbon can be made.

8.1.10. Sample Collection, Storage and Handling

See Appendix F for detailed explanation of collecting, storing and handling samples for nitrogen analysis.

8.2. Phosphorus

This section provides a detailed study on phosphorus and its effect on soil, including:

- Phosphorus loss pathways, including leaching, erosion and dissolution in stormwater.
- Forms of phosphorus (available, organic, inorganic) and chemistry, and dynamics, including availability.
- Phosphorus storage (phosphorus sorption) and its variability with soil type.
- Analysis methods – especially methods for measuring available P, phosphorus storage capacity and applicability of tests for different soil types.
- Analysis methods (standards, costs, accuracy, sample storage & handling).
- Review of Australian research on phosphorus (APL, Cattle and Beef CRC and others).
- Measuring phosphorus sustainability.

8.2.1. Loss Pathways Impacts of Phosphorus

Phosphorus is an essential plant nutrient. Holford (1997) states that most Australian soils are phosphorus deficient. However, attention in recent years has been diverted to the role of phosphorus in environmental water pollution. In conjunction with high nitrogen levels, elevated phosphorus concentrations in water resources (streams, lakes and dams) can cause eutrophication and excessive algal growths (Skerman, 2000). Thus, there is a need to minimise the loss of phosphorus from the soil, where it is beneficial, to groundwater and streams where it is harmful.

Skerman (2000) states that soil erosion and dissolution of soluble phosphorus in run-off water are the main phosphorus export paths. Consequently, the greatest phosphorus concentrations in streams and dams generally occur in deposited or suspended sediments. Under certain conditions, particularly anaerobic conditions in stratified lakes and dams, phosphorus may be released from the particulate matter into the water column where it is available for use by phytoplankton such as algae, blue-green algae and diatoms. Rapid increases in nuisance algal growth may occur under these conditions. In particular, high phosphorus levels have been linked to the occurrence of potentially toxic cyanobacteria (blue-green algae) blooms throughout Australia over the past decade.

Gardner *et al.* (1994) suggest that soil erosion is the main path for phosphorus exports to streams. Therefore, guidelines need to be framed to limit the slope of the reuse areas and the tillage strategy adopted when reusing effluent and manure on land.

At the Tulimba feedlot west of Armidale, in NSW, the surface and sub-surface nutrient and salt loss from feedlot manure application plots was investigated. The manure applications were 0 t/ha, 20–25 t DM/ha annually and 60 t DM/ha once at the start of the 3 year experiment. These treatments were compared with inorganic fertiliser application. Some of the manure application treatments were also supplemented with inorganic fertiliser (urea) to meet crop requirements.

This work identified the need to consider the soil's initial phosphorus status, buffering capacity and changes in phosphorus sorption through manure additions when calculating loading rates (Klepper 2001). It showed that leachate phosphorus losses in light soils (sandy loams) were directly related to available soil phosphorus concentrations. Thus, some soils have the potential to leak phosphorus before the soil reaches its storage capacity as defined by a phosphorus isotherm. Redding (2001) found that phosphorus accumulated 25 to 50 times that predicted by sorption isotherms after massive additions of effluent phosphorus. This demonstrated that soil storage of phosphorus is a reasonable concept, but the difficulty is in selecting an upper limit.

Manure applied either annually or in one larger application every few years decreased phosphorus sorption capacity. It was assumed this was because of phosphorus additions and not because of organic anions derived from the manure blocking the retention sites. It was concluded that repeated phosphorus additions expend the soil phosphorus storage capacity, providing a new starting point on the sorption curve when applying further phosphorus (Klepper 2001).

Manure applications produced the highest nutrient concentrations in surface and subsurface flows in between crops. Over two years, all treatments exceeded the nitrate nitrogen and phosphorus limits defined in ANZECC water quality guidelines as being detrimental to water health, although these concentrations would be diluted in streams. Furthermore, entrapment and sorption prior to this runoff reaching water bodies would reduce these concentrations in solution (Klepper 2001).

Recent APL-funded work by Redding investigated the occurrence of soluble organic phosphorus in effluent and sludge reuse areas and its mobilisation in water resources. The study showed that certain forms of effluent-phosphorus (particulate phosphorus) leach through some soils more readily than the forms of phosphorus contained in commercial fertilisers.

The study showed that effluent phosphorus could enter water resources through bypass leaching. Bypass leaching occurs where water moves down soil cracks, wormholes and root cavities. This prevents nutrients in the water from contacting and binding to the soil's surface. Bypass flow is most likely to occur when the surface of the soil becomes saturated. This highlights the need to avoid using effluent irrigation techniques that promote surface soil saturation, especially methods like flood and contour irrigation (Redding pers. comm., 2002).

Once effluent-P is sorbed to the surface of the soil, bypass flow becomes a much less important issue. The first few days after effluent application are critical for the sorption process. It is important to consider weather forecasting and apply effluent when there is minimum likelihood of rainfall within a few days of application. Effluent should not be applied to soil that is still wet from previous clean water irrigation or rainfall (Redding pers. comm., 2002).

8.2.2. Forms and Soil Dynamics of Phosphorus

Phosphorus is extremely chemically reactive, with more than 170 phosphate minerals identified. In all its natural forms (including organic), phosphorus is very stable or insoluble, and only a very small proportion exists in the soil solution at any one time (Holford, 1997).

Phosphorus exists in both inorganic and organic compounds in the soil. Inorganic compounds are mainly insoluble aluminium, iron and calcium compounds, and it is the orthophosphates (H_2PO4^-, HPO_4^{2-}, PO_4^{3-}) in the soil solution that are most available for plant uptake (Gardner *et al.*, 1994). Phosphorus is most available for plant uptake when in the unbound inorganic ortho-phosphate forms. A fairly small proportion of the phosphorus in the soil is in these forms.

The inorganic phosphorus is immediately available for both further plant uptake and chemical fixation reactions with soil minerals. The latter process is an immobilisation reaction since phosphorus is no longer readily available for plant uptake (Gardner *et al.*, 1994).

Organic phosphorus is contained in organic matter (humus, plant residues and manure) and is unavailable to plants in this form. However, it is slowly converted to orthophosphate over time. The conversion of organic phosphorus to orthophosphate by micro-organisms (the 'mineralisation' process), the desorption of orthophosphate from soil surfaces, and dissolution of inorganic phosphate compounds are the processes by which orthophosphates are made available for plant uptake from the soil solution. Organic material decomposition is driven essentially by microbial demand for carbon for new cell growth and only the phosphorus (and nitrogen) content in excess of microbial demand will become available for plant uptake and soil reactions. If the microbes produce excess inorganic phosphorus, there is a net phosphorus mineralisation. However, if all of the phosphorus is tied up in the microbial biomass, there is a net immobilisation of phosphorus. The balance between net immobilisation / mineralisation depends primarily on the carbon-nitrogen-phosphorus ratio of the material added to the soil. Some organic phosphorus is resistant to this breakdown and is routed into stable soil organic matter (Gardner *et al.*, 1994).

Roots have some capacity to collect non-solution phosphorus. All soils have some capacity to sorb phosphorus, ranging from low capacity in most sandy soils to high capacity in most clay soils. Therefore, only a small proportion of phosphorus is released into the soil solution each year.

The amount of phosphorus taken up by the plant and consequently exported from the site depends on the phosphorus content of the plant and its dry matter yield. All of the above plant ground material needs to be harvested and removed from the utilisation area to obtain the maximum phosphorus removal rate (Gardner *et al.*, 1994). The nutrient content of various crops is tabulated and discussed in Section 7.3.

Monitoring both the total phosphorus and the ortho-phosphorus concentration in the effluent provides a guide to the quantity of irrigated phosphorus that is readily available to the plants and the amount potentially available in the future.

The most effective methods of measuring available phosphorus (soil tests) are those that remove a proportion of labile phosphorus that is inversely related to buffer capacity. Soil tests that measure the concentration of phosphorus in solution actually measure availability rather than available phosphorus. Their efficacy on a range of soils depends on the uniformity of the soils' buffer capacities (Holford 1997).

8.2.3. Phosphorus Storage Capacity (P Sorption)

Inorganic phosphorus ions such as H_2PO4^-, HPO_4^{2-}, PO_4^{3-} can be adsorbed (sorbed) by soil minerals (a fast reaction) and fixed in the crystal lattice of soil minerals (a slow reaction) or precipitated as insoluble inorganic compounds in calcareous soils. A soil's phosphorus sorption capacity depends on the concentration of hydrous oxides of variable iron and aluminium (sesquioxides) and their variable charge characteristics. This sorption reaction is largely reversible.

Soil phosphorus sorption capacity varies widely, from low levels in sandy soils to high levels in strongly weathered clay soils e.g. red clay loams (oxisols). Table 33 shows the phosphorus sorption capacities for surface samples from a range of soils at a soil solution concentration of 0.5 mg of phosphorus per litre (Skerman, 2000).

The amount of phosphorus that can be fixed by a given soil is strongly related to inorganic phosphorus concentration in the soil solution. The relationship between the phosphorus sorbed and the solution concentration is called an *adsorption isotherm*.

To accurately assess the ability of a soil to sorb phosphorus a phosphorus sorption test (18 hour equilibration) needs to be applied. (Details on sampling and the test can be found in Appendix B).

Skerman (2000) states that the depth of soil to the base of the crop root zone should be considered the safe storage interval for applied phosphorus. To ensure that any phosphorus leached below the root zone does not adversely affect groundwater quality, it is recommended that the equilibrium solution concentration of phosphorus should not exceed 0.5 mg P/L at the base of the active root zone. Consequently, total net phosphorus applications (phosphorus application, minus phosphorus exported in plant material) should not produce active root zone solution concentrations exceeding 0.5 mg P/L.

TABLE 33 – PHOSPHORUS SORPTION CAPACITIES FOR SURFACE SAMPLES FROM A RANGE OF SOILS AT A SOIL SOLUTION CONCENTRATION OF 0.5 mg P/L. (SKERMAN, 2000)

Australian Soil Classification *and description*	**Great Soil Group**	**Soil Bulk Density, B** (kg/m^3)	**P Sorption Capacity S** (mg P/kg soil)	**P Sorbed per m depth P_T*** (kg P/ha)
Brown Sodosol - *Brown duplex with sodic sub-soil*	Soloths	1,300	50	650
Stratic Rudosol - *Poorly developed soil*	Podzol	1,500	45	675
Grey Vertosol – *Grey cracking clay*	Grey Clay	1,200	73	876
Black Vertosol - *Black cracking clay*	Black Earth	1,300	73	949
Brown Dermosol - *Brown non texture contrast soil*	Prairie Soil	1,200	102	1225
Brown Kandosol - *Brown non texture contrast soil*	Yellow Earth	1,300	142	1847
Brown Chromosol - *Brown duplex soil*	Yellow Podzolic	1,200	194	2330
Red Ferrosol – *Red volcanic soils*	Krasnozem	1,300	280	3641
Red Chromosol - *Red duplex soil*	Red Podzolic	1,200	304	3649

*P_T represents the total phosphorus sorption capacity and is equivalent to the total net amount of phosphorus that may be safely applied to the utilisation area over its effective lifespan.

The main problem with the soil test information currently available is that it has been considered from an agronomic perspective and not an environmental perspective (i.e. what is the minimum amount of phosphorus that needs to be applied to match the requirements of the crop). Investigating the problem from an environmental sustainability viewpoint (i.e. how much phosphorus can we apply to the soil before the practice is no longer sustainable) produces limited useful information. Generally piggery and cattle feedlot by-products reuse rates will be limited by the phosphorus application long before the nitrogen requirements of the plant are met. Thus, using the practice of applying phosphorus in excess of plant requirements and utilising soil phosphorus sorption capacity, the most appropriate test must ensure that there is no significant loss of phosphorus to the environment.

In Queensland, the amount of phosphorus that can be stored on reuse areas is calculated from the phosphorus sorption isotherm, at a soil solution concentration of 0.5 mg/L.

In NSW the common method of calculating safe phosphorus storage capacity is to estimate the total phosphorus sorption capacity by extrapolating the phosphorus sorption to a point where no further phosphorus is sorbed. The strength of phosphorus sorption is determined from the steepness of the curve at low phosphorus concentrations. For most soils, the phosphorus sorption strength is low to moderate, so only about a third of the total phosphorus sorption capacity can be sorbed to the soil before some leaching occurs. However, on some soils the sorption strength is much higher and up to one half of the total phosphorus sorption capacity may be applied before leaching occurs. When calculating the mass of phosphorus that can be sustainably applied to land, the total phosphorus sorption capacity before leaching occurs should be used (Kruger *et al*., 1995).

These levels and calculation methods have been reviewed by Redding (2002) as part of an APL funded project - Planning Safe Storage of Phosphorus in Soil. A review of this work is included below.

Observations and previously published literature, with MEDLI modelling was used to develop an improved method of estimating soil phosphorus storage capacities, improving on the previously published methodologies such as those in Skerman (2000). A number of assumptions were integral to the modelling approach taken.

The method described here by Redding (2002) is designed to produce generalised limits for effluent irrigation in a particular region. The results produced are very dependent on the climate, depth of soil and the crops produced. Where these factors significantly differ, more specific values should be used. In general, storage capacities estimated with this method are conservative, and this conservatism can be reduced through using more model parameters that exactly suit an enterprise. While MEDLI is used in this description, it is possible that some composite of other models may allow a similar process to be followed.

Soil phosphorus storage is not permanent. This is supported both by earlier research from Redding and by other published literature, most notably that of Barrow (1983). Phosphorus storage should be regarded as storage for future use, and therefore effluent reuse areas where an excess of phosphorus is to be applied, should meet the following criteria (in addition to those common to every area appropriate for effluent reuse):

- *The soils of the effluent reuse area should be suitable for on-going cultivation, preferably with a history of cultivation. If the soils are suitable for on-going cultivation, then there is a very high probability that phosphorus storage can be managed so that excess phosphorus is utilised before it is leached, even if the piggery closes and the land passes to other uses.*
- *The soil profile should be of a reasonable depth (e.g. in excess of 0.5 m in depth). The shallower the profile, the more difficult it is to manage reuse to prevent phosphorus leaching through the profile, and storage capacity in shallow profiles may be negligible. Good agricultural soils are likely to be reasonably deep.*
- *In situ grazing of effluent reuse areas is not an effective means for removing nutrients (Redding, et al. 2002). Where an area has a history of grazing, and is unsuitable for*

on-going cultivation, storage of excess phosphorus should not be considered. However, with careful management and monitoring of nutrient loads into and out of the system, it is possible to remove significant quantities of phosphorus in grazed dairy systems.

For each of the important soils of the effluent reuse areas of the district or region, follow the process below.

1. *Define a conservative upper limit to leaching. As a conservative estimate of storage capacity, use a 0.5 m depth of soil in the model runs. The storage capacity is later doubled to give a storage capacity per area, for 1 m depth of soil. Appropriate soil data should be entered into MEDLI's four layers, with a Layer 4 thickness of 5 mm (Technical/Soil Water menu item). MEDLI uses input parameters that allow for accumulation of excess phosphorus in the soil (the crop parameters and effluent application rates that are likely to occur and result in phosphorus application exceeding crop removal). Use a climate file that is realistically representative of the most leaching-conducive conditions of the district (relatively high rainfall, with lower evaporation, and 40 years or more of data). Annual phosphorus applications exceeding plant requirements should be realistic relative to current industry practice. The model should be allowed to run for long enough to allow some leaching below the soil profile. Determine the year in which Layer 4 sorbed phosphorus has increased by more than 1 mg/kg (select Run/Full for the data set. Select Output/Full from the menu, then from the output, clear the dialogue box then display sorbed phosphorus in Layer 4. View this data as text). This indicates that negligible phosphorus transport into this layer is predicted up to this date. Determine the exact date on which this level is first exceeded, then round to the nearest year (the indicator year). From the summary report of phosphorus leached from the profile, determine the average and standard deviation of annual leaching up to and including the indicator year. Calculate the leaching upper limit as follows: upper limit = average leaching + 2 x standard deviation*
2. *Defining the limit to storage in this profile. Using the summary output from MEDLI, determine the five-year average of leaching for each year. This should be a forward looking average, so the five year average for any single year will be the average of leaching values for that year and the four following. This prevents anomalous single year values from unduly influencing the evaluation, but does so in a conservative manner. The year for which the five-year average exceeds the upper limit (as calculated above) represents the year in which leaching may no-longer be considered negligible (the loaded year). Read off the profile load limit from the "Total Phosphorus Stored" (kg/ha) column that corresponds to the determined year.*
3. *Record the soil solution phosphorus concentrations for each of the soil layers at the end of the loaded year. To do this, view the full result set, and select the solution phosphate concentration for each of the 4 layers, then view the data as text.*
4. *Test for post-storage leaching. Modify the current MEDLI parameters to suit the leaching environment following effluent application. Firstly, reduce the pond drawdown depth to 0 m, so that no additional effluent is irrigated during the run (under the Enterprise/Pond menu option). Second, alter the soil solution phosphorus concentrations to those recorded in Step 3 (Technical/Soil Phosphorus). Finally, determine the land use that is most likely to be in place after effluent application (regardless of whether the piggery is still operational), and enter this into the MEDLI plant dialogue. Complete a summary MEDLI evaluation of the scenario (at least 40 years). If the five-year forward-looking leaching average (kg/ha/year) exceeds the upper limit calculated in Step 1 in any year, then return to Step 2, and choose the year prior to the current loaded year to replace it in the subsequent evaluation. This*

process should be repeated until storage in the profile does not produce profile leaching exceeding the upper limit in any year.

5. *Once a profile load limit that does not produce any subsequent leaching (kg/ha/0.5 m depth) has been determined, estimate the likely storage capacity for a 1 m soil profile by doubling this value (giving kg of P/ha/m depth). This is a conservative approach to creating a generalised value, since modelling a 1 m profile and using this as a "per metre of depth" guide is likely to allow leaching in profiles shallower than 1 m.*

Once this process is completed for each of the likely effluent reuse area soils for effluent-reuse in a particular district, it is possible to calculate a conservative generalised upper limit to soil solution concentration of phosphorus, based on the sorption curve, and the simplification that phosphorus is sorbed uniformly down the profile:

- *Convert the profile load limit to a sorbed mass of phosphorus per mass of soil (convert units as necessary), Profile load limit [mg] / (10000 [m2/ha] x depth [m] x bulk density [kg/m3]).*
- *Use the value calculated to read the solution concentration off the x-axis.*
- *Choose the lowest value from the list of soils modelled for the district and use this as a rough guide to likely storage capacity. This value will vary greatly with climate (Table 9).*

TABLE 34 – ESTIMATED PHOSPHORUS SOIL STORAGE PARAMETERS FOR TOOWOOMBA AND WAUCHOPE CLIMATE DATA (REDDING, 2002)

	Leaching upper limit	Profile load limit	Toowoomba Solution concentration limit	Wauchope Solution concentration limit
	(kg/ha/yr)	(kg/ha)	(mg/L)	(mg/L)
Soil 1	0.17	4943	3.70	0.18
Soil 2	0.06	2713	2.67	0.90
Soil 3	0.12	2743	4.61	0.30
Soil 4	0.18	282	3.04	4.60

Note: Annual excess soil phosphorus loading is approximately 90 kg/ha during the loading cycle, MEDLI forage sorghum crop rotation being selected for the post loading leaching cycle for Soils 1,2 and 3, and broccoli for soil 4.

It is recommended that storage of phosphorus be allowed, based on the calculated storage capacity from the phosphorus sorption isotherm, at a soil solution concentration of 0.5 mg/L. However, this soil solution concentration level requires review, pending the findings of the recent Redding work. It would be possible to generate appropriate soil solution concentration levels for different soil types and regions from currently available data. A reuse area should be used to store phosphorus only if it is good cropping land and providing a plan is in place to continually crop the area after effluent or solids reuse has ceased to remove the stored phosphorus as it is released.

8.2.4. Analysis Methods

Most unfertilised Australian soils contain small amounts of total phosphorus (P), usually less than 0.2%, with much immobilised in forms not readily available to plants, such as organically bound phosphorus and insoluble mineral phosphorus. The quantity of phosphorus available to plants is seldom related to the total reserves of this essential element (Rayment and Higginson, 1992).

In Australia, several empirical extractants for soil phosphorus are employed to determine the amount of "available" phosphorus. They are the bicarbonate extractions of Colwell (1963) and Olsen *et al.* (1954), lactate-extractable P, fluoride-extractable phosphorus (Bray 1), dilute $CaCl_2$-extractable phosphorus and acid-extractable phosphorus " (Rayment and Higginson, 1992). Other methods include the BSES method described in Moody and Bolland (1999).

Moody and Bolland (1999) warn that in some calibration studies, although the phosphorus test may be described by one of the above methods, the original extracting conditions have been altered. These details should be checked before comparing critical values for the 'same' soil test from different sources. There is no one single relationship to interpret between phosphorus soil tests. Moody and Bolland (1999) noted a few studies have developed relationships between some of the phosphorus soil tests so that, for a given group of soils, one test can be used to predict another.

See Appendix B for full details on analysis methods.

8.2.5. Measuring Phosphorus Sustainability – Soil Indicator for Phosphorus

Moody and Bolland (1999) have developed a generalised table for Colwell-extractable phosphorus levels to provide a guide to soil phosphorus status. Table 35 was generated from generalised interpretation guidelines used by various Australian State Departments of Agriculture.

TABLE 35 – GENERALISED INTERPRETATION GUIDELINES FOR COLWELL EXTRACTABLE PHOSPHORUS (0-10 CM) – MOODY AND BOLLAND (1999).

		Crop Demand		
Soil phosphorus status	**Soil phosphorus sorption category**	**Low (e.g. dryland pasture)**	**Moderate (e.g. grain crops)**	**High (e.g. vegetable crops)**
Low	Low	< 10	< 15	< 20
	Moderate-high	< 20	< 30	< 50
Medium	Low	10 – 30	15 – 45	20 – 60
	Moderate-high	20 – 60	30 – 90	50 – 150
High	Low	> 30	> 45	> 60
	Moderate-high	> 60	> 90	> 150

Skerman (2000) states that significant leaching of phosphorus generally occurs only when the soil is heavily overloaded with phosphorus. Table 36 suggests levels of available phosphorus concentrations in surface soil that will meet plant requirements and should not result in significant losses to surface water, provided runoff is controlled via good design and management. These limits are commonly exceeded in normal agricultural soils and are suggested as a trigger for further investigation by comparison with analysis results for 'virgin' soils receiving no effluent or manure or if there are doubts about the sustainability of the reuse practice. The limits used in Table 36 are derived from field measurements of a set of soils, where the numbers are the mean value, plus one standard deviation for each category. They do not generally apply to vertosols, as they may have high levels of available phosphorus in their 'virgin' state. Site-specific, background available phosphorus levels are likely to be required for these soil types.

A method of deciding the vulnerability of a reuse site in terms of phosphorus export in runoff and erosion is to obtain extractable phosphorus levels using the most appropriate phosphorus extraction for the soil type on an area that has not received effluent and solids to obtain baseline data. This should be compared against annual monitoring of the reuse area for extractable phosphorus levels to evaluate trends.

The soil profile to the base of the crop root zone should be considered the safe storage interval for applied phosphorus to avoid phosphorus leaching. To prevent excessive leaching of phosphorus below the root zone, it is recommended that the equilibrium solution concentration of phosphorus of 0.5 mg P/L be used to estimate the safe phosphorus storage capacity. Thus, phosphorus applications exceeding removal by the plant material should not go beyond the phosphorus sorption capacity of the soil at an equilibrium solution concentration of phosphorus of 0.5 mg P/L. However, this soil solution concentration level needs review pending the findings of the recent Redding work. It would be possible to generate appropriate soil solution concentration levels for different soil types and regions from currently available data.

A reuse area should be used to store phosphorus only if it good cropping land and providing a plan is in place to continually crop the area after effluent or solids reuse has ceased to remove the stored phosphorus as it is released. The phosphorus storage capacity of the reuse area should also be determined by measuring a P sorption isotherm every five years.

The P sorption capacity of the soil will generally change down the soil profile due to decreasing levels of available P and changes in soil texture. Phosphorus sorption capacity can be determined by a single average test of the soil profile to the base of the root zone to reduce significant analysis costs. However, it may be beneficial for producers to test the P sorption capacity of different soil layers in some instances.

TABLE 36 – SUGGESTED UPPER LIMITS FOR AVAILABLE PHOSPHORUS IN TOPSOIL (SKERMAN, 2000).

Clay Content	pH	Colwell phosphorus (mg/kg)
less than 30%	less than 7	31
less than 30%	greater than 7	59
greater than 30%	less than 7	75
greater than 30%	greater than 7	85

Note: These levels do not apply to some soils, e.g. black vertosols.

Table 36 could be further enhanced by some further work on the study conducted by Burkitt *et. al*. (2002), who collated Colwell extractable phosphorus for a large range of Australian soils.

Despite the large amount of research into soil phosphorus chemistry, it is still unclear which is the best phosphorus soil test for particular soils. There is considerable data on Colwell P, but some researchers suggest that the Colwell test may not be the best test of potentially available phosphorus in some soil types (particularly some acid soils of NSW), with acid extractable tests being more appropriate for these soil types. Analytical data indicates that the exchangeable acidity of even extremely acid soils (e.g. pH 2.7, Redding, 1997) is not sufficient to neutralise a significant proportion of the dissolved bicarbonate in the extracting Colwell solution (Redding pers. comm., 2002).

The Department of Land and Water Conservation (NSW), Soil And Land Information System (SALIS) database ranks various chemical test results for NSW soil tests, including Bray P (derived from Abbott (1985). These rankings are shown in Table 37. The high ranking of 20-25 mg/kg Bray P in the surface soil could be used as a guideline measure of a trigger for further investigation. This further investigation could include comparison against background data.

TABLE 37.- CHEMICAL TEST RESULT RANKINGS FOR BRAY PHOSPHORUS (mg/kg)

Very Low	Low	Moderate	High	Very High
<5	5-10	10-20	20-25	>25

Redding pers. comms. (2002) developed limits of available phosphorus in the surface soil for the BSES method, based on the same principles as the limits for Colwell (mean + one standard deviation) depending on the level of clay. These are shown in Table 38. It should be noted that these numbers are derived from a relatively small data-set and may need refining when more data is available.

TABLE 38 - BSES PHOSPHORUS (mg/kg) GUIDELINE LEVELS

Clay Content	Average	Standard Deviation	Guideline
less than 30%	17	14	31
greater than 30%	59	72	131

Both the Bray and BSES may be more appropriate measures of available P in certain soils (e.g. acid).

To investigate any possibility of P leaching, particularly with sandy soils, measurement of available P levels at 50 – 60 cm (or the base of the root zone) is also suggested.

The knowledge of phosphorus storage capacities of different soils is still limited and individual site assessments are still likely to be required.

Another test that offers potential is the simple test for estimating phosphorus buffer capacity (PBC) that was developed by Burkitt *et al.* (2002). Their methods provide a simple and accurate method for estimating PBC. However, this work requires further evaluation to ascertain whether their data can be used to provide simple indices for determining phosphorus sustainability of a range of soil types, not only in NSW, but for the cropping soils of Australia in general.

More detailed information on phosphorus and in particular which is the most suitable phosphorus test can be found in Appendix B.

8.2.6. Sample Collection, Storage and Handling

See Appendix F for full details on the collection, storage and handling of samples in relation to phosphorus.

8.3. Salinity

This section includes a detailed study on salt and its effect on soil. It examines how reusing the effluent and solid by-products of intensive livestock production affects soil, surface water and groundwater salinity. The following is covered in detail:

- Definition of salinity.
- Salinity effects on the environment.
- Measuring salinity (TDS, EC).
- Sample collection, storage & handling.
- Interpreting salinity results.
- Managing salinity problems, including the leaching of salts (release to the environment) in order to maintain the productive capacity of the land.

8.3.1. Definition of Salinity

Salinity refers to the total dissolved solids in a liquid or in a soil solution. Salts are mostly added to the soil through soil formation, hydrologic processes and rainfall (Shaw 1999). Reusing effluent and solid by-products from piggeries and cattle feedlots adds significant quantities of salt to the soil.

A saline water is one containing sufficient soluble salts to affect plant productivity under specific environmental and management conditions (Shaw and Dowling, 1985). In groundwater, most salt is typically sodium chloride (NaCl), with varying amounts of calcium sulfate ($CaSO_4$, gypsum), sodium sulfate (Na_2SO_4, soda ash), sodium bicarbonate ($NaHSO_4$, baking soda) and magnesium sulfate ($MgSO_4$, epsom salts) (Salt Action, 2000). However, the total salt load in effluent and solid by-products from piggeries and cattle feedlots comprises plant nutrients including nitrates, phosphates and sulfates; soil amendments including calcium and organic matter; and sodium chloride. Almost all of the potassium in effluent is the K^+ ion in the form of a salt (Zhang and Hamilton, N.D.). Because of their potential detrimental effects on plant growth it is the sodium and chloride that are of most interest here. Effluent generally has a relatively high ratio of sodium to the other cations. This is detailed further in section 8.4.

A saline soil contains sufficient soluble salts within the profile to reduce plant productivity (Shaw and Dowling, 1985). Soil salinity can arise on reuse areas when the amount of salt added by the effluent or solid by-products is not balanced by salt removal through leaching below the plant root zone. Some individual salts (e.g. chloride, sodium and boron) may cause toxic effects through ion accumulation in the leaves causing leaf damage.

The most common salts in Australian soils contain sodium and chloride ions. However, magnesium, calcium and sulfate ions may also occur in the lower soil profile (Charman and Wooldridge, 2000). The solubility of common salts formed from these ions is given in Table 39.

Sodicity is a high proportion of exchangeable sodium ions relative to the total exchangeable cations in soil or water (including effluent). Salinity and sodicity can occur separately, or together, in soils. These are discussed separately below.

TABLE 39 – SOLUBILITY OF COMMON SALTS

Salt	Formula	Solubility (mmole/L)*
Calcium carbonate	$CaCO_3$	0.5
Magnesium carbonate	$MgCO_3$	2.5
Calcium bicarbonate	$Ca(HCO_3)_2$	3-12
Magnesium bicarbonate	$Mg(HCO_3)_2$	15-20
Calcium sulfate	$CaSO_4.2H_2O$	30
Sodium sulfate	$Na_2SO_4.1OH_2O$	683
Sodium bicarbonate	$NaHCO_3$	1642
Magnesium sulfate	$MgSO_4.7H_2O$	5760
Sodium chloride	NaCl	6108
Magnesium chloride	$MgCl_2.6H_2O$	14,955
Calcium chloride	$CaCl_2.6H_2O$	25,470

* Solubility of carbonate minerals depends on the concentration of carbon dioxide in the solution and soil air. (Doneen 1975, cited by SalCon (1997)).

8.3.2. Salinity Effects on the Environment

Salts added to the soil accumulate when there is preferential loss of water by evaporation or evapotranspiration, rather than by drainage. Consequently, salt accumulates in all but the most permeable soils. In the presence of shallow groundwater systems, salts tend to accumulate in the upper soil layers. Where water tables are at least 2 m below the soil surface, salt tends to accumulate at the base of the active root zone or at the depth of effective soil wetting. The extent of salt accumulation in the soil depends on the permeability of the soil (which influences leaching), the presence and type of vegetation (evapotranspiration) and the amount and seasonal distribution of rainfall. In high rainfall areas, soils have low salt accumulation because leaching is sufficient to remove surplus salts (Shaw 1999).

Addition of salt to land areas is an environmental issue for the following reasons.

1. Soil salinity can reduce plant growth and yields through dehydration. This happens because the dissolved salts lower the potential for water to pass into the roots. Yields can decline by 20-30% before the signs of salinity are obvious (Salt Action, 1999). Crops may also appear to be water-stressed even when supplied with adequate water. However, the effect is often more obvious in dry years (SalCon, 1997).

2. If crop yields are reduced significantly, bare soil patches may form. This significantly increases the risk of soil erosion.

3. Different plants have differing abilities to take up saline water. Hence, the soil salinity influences the crops that can be grown and the composition of pastures. Excess levels of specific salts may also be a problem. For instance, an excess of sodium or chloride accumulates in plant leaves producing burning, necrotic patches and defoliation. An associated effect is a reduction in the availability of calcium and magnesium, which may produce deficiency symptoms. An excess of boron is expressed through yellowing of the margins, crumpling, blackening and leaf distortion (SalCon, 1997).

4. A high salt or sodium concentration can degrade soil structure, cause scalding and significantly increase erosion. The soil may appear fluffy and light under highly saline conditions (SalCon, 1997).

5. Salts leaching through the soil may reduce the quality of underlying groundwater.

6. Saline runoff and soil erosion may reduce the quality of receiving surface waters.

7. Highly saline soil solutions may mobilise heavy metals and other potentially toxic substances in the soil (Charman and Wooldridge, 2000).

8.3.3. Measuring Salinity

Electrical conductivity (EC) and total dissolved solids (TDS) are the most common measures of salinity. Total dissolved ions (TDI) is another measure.

EC measures the quantity of electricity conducted by a liquid. It is the reciprocal of electrical resistance and increases with salt concentration. The standard unit is deci-Siemens per metre (dS/m). Since many salts disassociate to the ionic form in water, measuring the EC of liquids or soil solution provides a good measure of the total salt concentration (SalCon, 1997). A problem with EC as a measure of effluent salinity is that it includes all the dissolved solids, including nutrients needed for plant growth. It does not provide a good measure of the harmful salts (particularly sodium and chloride).

The EC of water can be determined in a laboratory or in the field using a portable EC meter. If field meters are used, these need to be calibrated and need to have a temperature compensated probe since EC is temperature dependant (SalCon, 1997).

For soils, the common laboratory measurement methods for EC are 1:5 soil water suspension, soil saturation extract and electrical conductivity of soil at measured or maximum field content.

The $EC_{1:5}$ method was developed to overcome some of the difficulties in using the saturation paste method with heavy-textured Australian soils. However, to relate EC measurements to plant salt tolerance data, soil leaching and soil behaviour, the data must be in the EC_{se} form. $EC_{1:5}$ conversion to EC_{se} is likely to produce a less accurate result than direct measurement of EC_{se}. However, because EC_{se} is an imprecise measure and a difficult technique, a general prediction of EC_{se} from $EC_{1:5}$ may be the most appropriate measure (Shaw, 1999).

A range of formulae are available for converting $EC_{1:5}$ to EC_{se}. The SALF software available from Department of Natural Resources and Mines (DNRM) (Queensland) includes a SALFCALC component that readily converts between EC methods at different salinities, based on soil properties (SalCon, 1997). However, a simple extrapolation from texture is given in Shaw (1999). This is presented as Figure 3, with clay content of soil based on the data presented in Table 40.

FIGURE 3 – RELATIONSHIP BETWEEN EC_{se} AND MEASURED $EC_{1:5}$

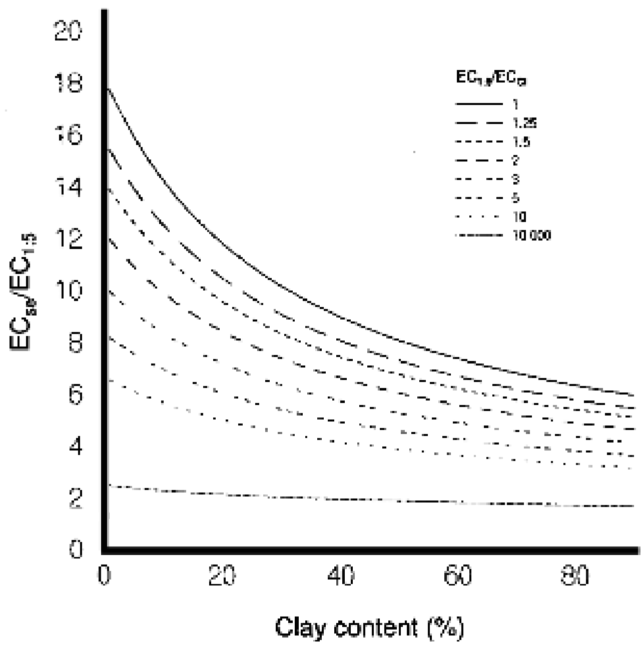


TABLE 40 – RELATIONSHIP BETWEEN TEXTURE CLASS, TEXTURE GRADES AND CLAY CONTENT

Texture Class	Texture Grades of McDonald & Isbell (1990)	Median Clay Content, approx. % (Shaw, 1994)
Sand	Sand	5
Loamy sand	Loamy sand, clayey sand	7
Sandy loam	Sandy loam	15
Silty loam	Loam, silty loam	25
Clay loam	Clay loam, silty clay loam	32
Light clay	Light clay, light medium clay	40
Medium clay	Medium clay	50
Heavy clay	Heavy clay	65

(Cited by Shaw, 1999).

EC_{se} determined by analysis or calculated from $EC_{1:5}$ is the recommended measure of soil solution salinity. The SalfCalc model can be used to convert $EC_{1:5}$ to EC_{se}. Laboratory methods for salt measurement should be undertaken according to Rayment and Higginson (1992).

Because plants respond to salinity throughout the root zone, it is useful to convert EC_{se} at a number of depths to a single value representing the entire root zone. Root zone salinity is commonly expressed as either the average root zone salinity or the water uptake weighted root zone salinity. Both methods require an estimate of the root depth of the particular plant being grown.

Average root zone salinity is the sum of the salinity measurements for a series of root zone depth increments divided by the number of root depth increments. A more realistic predictor of plant response to salinity is water uptake weighted root zone salinity, which is based on the actual water uptake pattern of plants. For effluent reuse areas, and other irrigated land, the conversion of EC_{se} to water uptake weighted root zone salinity is recommended.

Water uptake weighting patterns at 0.1 m increments for three common rooting depths are given in Table 41. The actual EC measurement at each depth is multiplied by the weighting factor for the root zone depth of interest and the values summed to find the water uptake weighted root zone salinity (SalCon, 1997). Table 42 provides water uptake weighting pattern factors for standard survey depths and three common rooting depths.

TABLE 41 – WATER UPTAKE WEIGHTING PATTERN FACTORS (WUW) FOR 0.1 m DEPTH INCREMENTS FOR THREE COMMON ROOT ZONE DEPTHS

Soil increment (m)	Weighting factor for each 0.1 m increment where root zone depth is:			Analysed EC_{se} (dS/m)	Weighted EC_{se} (dS/m) (EC * weighting factor for 0.9 m deep soil)
	0.6 m	0.9 m	1.2 m		
0-0.1 m	0.35	0.27	0.23	0.4	0.10
0.1-0.2 m	0.18	0.14	0.12	0.4	0.06
0.2-0.3 m	0.15	0.11	0.10	0.4	0.05
0.3-0.4 m	0.13	0.10	0.08	0.5	0.05
0.4-0.5 m	0.11	0.09	0.07	0.7	0.06
0.5-0.6 m	0.08	0.08	0.07	1.1	0.09
0.6-0.7 m		0.08	0.07	1.9	0.15
0.7-0.8 m		0.07	0.06	3.2	0.22
0.8-0.9 m		0.06	0.06	4.2	0.25
0.9-1.0 m			0.06	average	wuw
1.0-1.1 m			0.05	root zone	- sum of
1.1-1.2 m			0.03	Mean	Values
SUM	1.0	1.0	1.0	=1.42	=1.03

(Shaw *et al.* 1987).

TABLE 42 – WATER UPTAKE PATTERN WEIGHTING FACTORS FOR STANDARD SURVEY DEPTHS AND THREE COMMON ROOTING DEPTHS

Soil increment (m)	Weighting factor for each 0.1 m increment where root zone depth is:		
	0.6 m	0.9 m	1.2 m
0-0.1 m	0.35	0.27	0.23
0.2-0.3 m	0.46	0.35	0.10
0.5-0.6 m	0.19	0.25	0.07
0.8-0.9 m	-	0.13	0.06
1.1-1.2 m	-	-	0.03
SUM	1.0	1.0	1.0

(based on Shaw *et al.* 1987).

For effluent reuse areas, the conversion of EC_{se} to water uptake weighted root zone salinity is recommended.

Salinity can also be measured by total dissolved solids (TDS) or by total dissolved ions (TDI). TDS measures the mass of total dissolved solids per unit volume. It can be measured by evaporation or can be calculation. TDI is the sum of the analysed cations plus anions expressed on the basis of mass per volume. The ions considered must include at least Ca^{2+}, Mg^{2+}, Na^{+}, CO_3^{2-}, HCO_3^{-}, SO_4^{2-} and Cl^{-}. (This is also the measure of Total Soluble Salts (TSS)) (SalCon, 1997). The TDI method is inferior to TDS since it excludes total silica and does not account for the conversion of HCO_3^{-} to CO_3 on evaporation. As TDS and TDI include a range of beneficial ions these measures overstate the quantity of harmful salts (sodium and chloride).

A rule of thumb for converting TDS mg/L to EC (dS/m) is division by 640. This is not a valid relationship for effluent and will vary between effluents of different quality. Because it is easier to measure, $EC_{1:5}$ which can then be converted to EC_{se} is the preferred measure of salinity.

Sodium chloride is the salt of most interest in reuse areas, since it is the salt most likely to cause harm. Sodium measurement is covered in section 8.4. Chloride can be measured using the 1:5 soil/water method. Critical levels for salinity are 120 mg/kg for sands to sandy loams, 180 mg/kg for loams to clay loams and 300 mg/kg for clays. Levels exceeding these concentrations may cause salinity damage depending on plant tolerance and soil drainage (Hughes *et al.,* ND).

If further investigations are warranted, the soil Na^{+} + Cl^{-} concentration should be determined for the reuse and background sites since sodium chloride is the main salt of interest from a soil degradation perspective. The soil Na^{+} + Cl^{-} concentration of the soil should be less than 150% of background levels at 50-60 cm (or base of root zone).

See Appendix C for full details on measuring salinity.

8.3.4. Sample Collection, Storage and Handling

Samples for laboratory analysis of EC need to be dispatched as soon as possible after collection. Delays and high temperatures change the EC by precipitating salt out of solution. Sample bottles should be filled completely to exclude air (SalCon 1997).

For more information see Appendix F.

8.3.5. Interpretation of Salinity Results

Both effluent quality and soil parameters should be examined when assessing the suitability of effluent for reuse. Most water classification methods are based on EC_{se} or $EC_{1:5}$. Table 43 presents irrigation water salinity classes from Gill (1984). These are based simply on irrigation water EC_{se}.

Table 44 gives irrigation water classes developed by Shaw *et al.* (1987) and presented in SalCon (1997). This table uses similar EC_{se} categories to the earlier Gill (1984), but also considers chloride in ranking water quality.

TABLE 43 – SALINITY CLASSES FOR IRRIGATION WATERS

Irrigation Water Quality (EC_{se})	Water Salinity Class	Plant Salt Tolerance Groupings
<0.65	1	Suitable for all crops except tobacco (if chloride < 25 mg/L, delete "except tobacco").
0.65-1.3	2	Suitable for all except very low salt tolerant crops.
1.3-3.0	3	Suitable for medium and high salt tolerant crops only.
3.0-5.0	4	Suitable for high salt tolerant crops only.
5.0-8.0	5	Generally unsuitable unless the soils are permeable and crops are very high salt tolerant.
>8.0	6	Too saline for irrigation.

From Gill 1984.

TABLE 44 – IRRIGATION WATER QUALITY CRITERIA FOR SALINITY BASED ON 90% YIELD OF PLANT GROUPS OF MAAS & HOFFMAN (1977)

Irrigation Water Quality (Assume LF = 0.15)		Water Salinity Rating	Plant Salt Tolerance Groupings
EC (dS/m)	Chloride (mg/L)		
<0.65	<220	Very low	Sensitive crops
0.65-1.3	220-440	Low	Moderately sensitive crops
1.3-2.9	440-800	Medium	Moderately tolerant crops
2.9-5.2	800-1500	High	Tolerant crops
5.2-8.1	1500-2500	Very high	Very tolerant crops
>8.1	>2500	Extreme	Generally too saline

(Table presented in SalCon 1997, based on criteria developed by Shaw *et al.* 1987).

For agricultural crops, salt tolerance is the ability of plants to survive and produce economic yields under saline conditions (Maas and Hoffman, 1977). Unfortunately, laboratory methods for measuring soil salinity do not readily relate to plant performance. The actual salt tolerance of different species varies depending on:

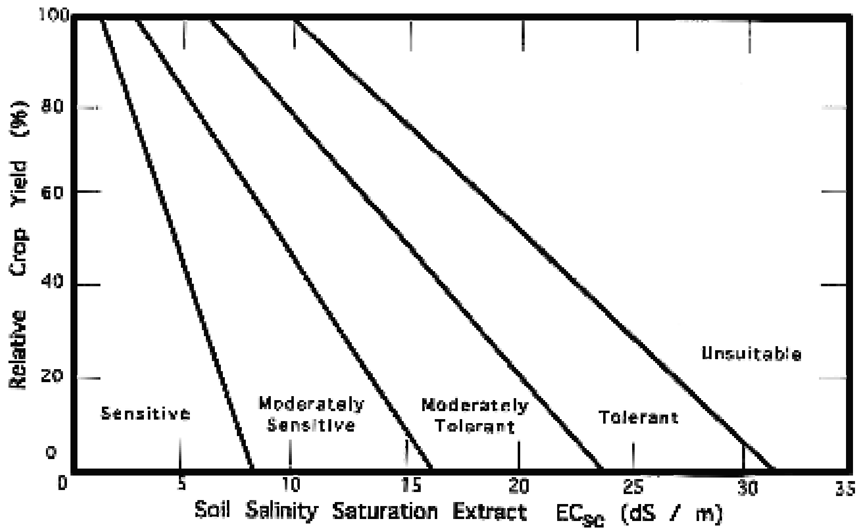
- The distribution of salt in the soil profile.
- Management practices e.g. some crops can tolerate higher salinity in the soil than in water applied to leaves.
- Climate e.g. plants tolerate salt better when the weather is cool and humid.
- Soil fertility, which may determine whether salinity is the primary limitation to growth (Shaw 1999).

However, plants usually respond to total salinity as an osmotic effect. They usually have a threshold salinity level, beyond which there is an approximately linear decrease in relative yield with increasing salinity.

The USSL (1954) scheme is a universally accepted scheme that uses plant salt tolerance as a basis for salinity assessment. The salinity classes apply if the EC_{se} anywhere in the root zone reaches the specified level. However, this scheme is unsuitable for Australian soils that are much less permeable than their USA counterparts, and have higher salt accumulation rates at depth (USSL (1954), cited by SalCon (1997)).

Maas and Hoffman (1977) slightly revised the USSL (1954) assessment criteria. The relationship they derived between soil root zone EC_{se} and relative plant yield, along with five salt tolerance divisions, is shown in Figure 4. The plant salt tolerance criteria are based on average root zone salinity for plants grown under high leaching fractions. This method accounts for variations between plant species, but does not consider the effect of soil texture (Maas and Hoffman (1977), cited by SalCon (1997)).

FIGURE 4 – RELATIVE CROP YIELD IN RELATION TO SOIL SALINITY (EC_{se}) FOR PLANT SALT TOLERANCE GROUPINGS REDRAWN FROM MASS & HOFFMAN (1977) BY SHAW (1999)



Northcote and Skene (1972) attempted to make the USSL (1954) scheme more relevant to Australian conditions by including texture and depth terms. The method is based on the chloride content of a 1:5 soil water suspension approximating an EC_{se} of 4 dS/m of USSL (1954). However, chloride alone underestimates salinity if gypsum or sodium carbonates are present. Depth is restricted to 1 m (Northcote and Skene (1972), cited by SalCon (1997)).

Shaw *et al.* (1987) developed a salinity tolerance classification scheme based on Maas and Hoffman (1977) that classifies plant species by the salinity at which a 10% yield reduction occurs. It includes five soil salinity ratings (an additional "very low" salinity class was added). Average root zone salinity, coupled with four soil texture classes, or water uptake weighted salinity can be used with this scheme. This is shown in Table 45.

TABLE 45 – SOIL SALINITY CRITERIA AS EC_{se} CORRESPONDING TO A 10% YIELD REDUCTION FOR THE PLANT SALT TOLERANCE GROUPINGS OF MAAS & HOFFMAN (1977) AND THE EQUIVALENT $EC_{1:5}$ FOR FOUR RANGES OF SOIL CLAY CONTENT

Plant Salt Tolerance Grouping	EC_{se} Range (dS/m)	Corresponding $EC_{1:5}$ Based on Soil Clay Content (dS/m)				Soil Salinity Rating
		10-20% clay	20-40% clay	40-60% clay	60-80% clay	
Sensitive crops	<0.95	<0.07	<0.09	<0.12	<0.15	Very low
Moderately sensitive crops	0.95-1.9	0.07-0.15	0.09-0.19	0.12-0.24	0.15-0.3	Low
Moderately tolerant crops	1.9-4.5	0.15-0.34	0.19-0.45	0.24-0.56	0.3-0.7	Medium
Tolerant crops	4.5-7.7	0.34-0.63	0.45-0.76	0.56-0.96	0.7-1.18	High
Very tolerant crops	7.7-12.2	0.63-0.93	0.76-1.21	0.96-1.53	1.18-1.87	Very high
Generally too saline for crops	>12.2	>0.93	>1.2	>1.53	>1.87	Extreme

(Shaw *et al.* 1987).

Rengasamy and Bourne (1997) suggest that sandy or loamy soils are generally saline if the $EC_{1:5}$ exceeds 0.4 dS/m while clay soils are generally saline if the $EC_{1:5}$ exceeds 0.7 dS/m. However, they acknowledge that the critical EC values vary between crops.

Foale (1998) provides a simple classification of $EC_{1:5}$ classes and crop yield depression. His data is presented in Table 46.

TABLE 46 – SALINITY CLASSES AND LIKELY CROP YIELD DEPRESSION (FOALE, 1998)

$EC_{1:5}$	Yield Depression
Below 0.2 dS/m	Very low
0.2-0.5 dS/m	Low
0.5-0.7 dS/m	Medium
Above 0.7 dS/m	High

This classification does not account for differences in salt tolerance between crops and does not consider the effect of soil texture.

It is suggested that the Shaw *et al.* (1987) classification scheme be used to assess the salinity of Australian soils. It considers soil texture and can be used with average root zone salinity (with soil texture) or water uptake weighted salinity data.

An extensive list of plant salt tolerance data for a range of crops is provided as Appendix F (taken from SalCon, 1997). These data provide the salinity threshold at which any yield reduction would occur (salinity threshold) plus the salinity at which a 10% yield reduction would be expected. These data are not absolute and vary with:

1. Root zone salinity – these data have mainly been obtained from laboratory trials where salinity is the only variable examined. This ignores the interactions between soil salinity and other factors that occur in the field.

2. Stage of plant growth – plant species vary in their tolerance to salinity depending on their stage of growth. For some plants, salt tolerance is highest in the early stages of growth. For other plants, the reverse is true.

3. Management practices – some plants can tolerate higher salinity in the soil than in irrigation water. Irrigating below the leaves and at night reduces plant leaf susceptibility to saline irrigation water.

4. Climate – plants are more tolerant of salinity under cool or humid conditions.

5. Soil fertility – soil fertility has a variable effect on plant tolerance to salinity (Shaw, 1999).

Appendix E provides conversion factors for EC and TDS units.

8.3.6. Managing Salinity

This section provides tools to help prevent or manage salinity in reuse areas. Soil salinity for effluent reuse areas must be managed through appropriate reuse practices and leaching. The salinity of solid by-products is rarely a concern, unless these are applied at extremely high rates (Hicks and Hird, 2000).

The threshold salinity data presented in Appendix E relates crop response to salinity in the root zone. The analysed TDS content of effluent or manure and a knowledge of soil properties can be used to calculate the approximate increase in salinity from adding effluent or manure at given rates. For instance:

1. Calculate salt load in effluent by multiplying quantity of effluent (e.g. 200,000 L/ha/yr) by its salt content (e.g. 3200 mg/L), which gives 620 kg salt/ha/yr.

2. Calculate volume of soil depth of interest by multiplying soil depth (e.g. topsoil 10 cm deep) by 10,000 m^2/ha, which gives 1000 m^3/ha.

3. Calculate volume of soil water by multiplying soil water content at field capacity (e.g. 50% v/v) by the soil volume from the previous step (1000 m^3/ha), which gives 500 m^3 (NOTE: it is important to use a reasonable estimate of soil water content at field capacity. A light textured soil may have a soil moisture content at field capacity of 15-25% v/v, which has a significant bearing on the calculation.)

4. Calculate salinity of soil solution by dividing mass of salt applied (640 kg/ha from step 1 * 1000) by soil water volume (500 m^3 from step 3), which gives 1280 mg/L.

5. Convert this salinity (1280 mg TDS/L) to an approximate EC by dividing by 640, giving 2.0 dS/m. Because nutrients in effluent (particularly potassium) contribute very significantly to EC, this simple calculation overstates the quantity of harmful salts added to reuse areas. Hence, this calculation method must be considered very conservative. Nevertheless, it is useful for management since it overstates rather than underestimates the soil salinity.

6. Convert the resulting EC (2.0 dS/m) to approximate EC_{SE} by dividing by two (as a rule of thumb, saturated soil water content is about double the moisture content at field capacity), giving 1.0 dS/m.

NOTE: this process can be used to estimate manure addition by solid by-product reuse by substituting the mass of solids for the volume of effluent and the salt content of the solids (g/kg d.b.) for the effluent salt concentration in step 1.

The calculated value can be added to the initial soil EC_{se} to estimate the EC_{se} after effluent or manure additions. This can then be compared with the threshold salinity for crops that might be grown on the reuse area. It is important to realise that this method does not consider leaching losses that might reduce the EC_{se} of the soil. Hence, the output is conservative (based on: Gardner *et al.* 1995). The SALF program can be used to estimate the change in soil EC after effluent irrigation. It accounts for leaching by considering the leaching fraction of the rainfall and irrigation applied.

To prevent salt from accumulating in soil, salt additions through effluent irrigations must be matched by salt removal through leaching below the root zone. The proportion of applied water that must drain through the root zone to maintain soil salinity at acceptable levels is called the leaching fraction (LF). USSL (1954; cited by SalCon, 1997) use the term leaching requirement (LR) instead. It is calculated by the formula:

$LR = c_i / c_o = Q_o / Q_i$

where

Q_i is the depth of water entering the soil
Q_o is the depth of water draining from the soil
c_i is the EC of water entering the soil
c_o is the EC of water draining below the soil (salinity of deep drainage D_d).

This formula is widely used for long-term equilibrium situations, but is only correct if there is no salt precipitation or ion exchange within the root zone. Q_i and c_i are readily measured. If c_o is in equilibrium with the salt concentration of a soil close to field capacity, then Q_o can be estimated (SalCon, 1997).

Water drainage below the root zone is estimated by relating salt concentration at depth to the salt concentration of water inputs (rain plus irrigation water), weighted according to relative volume. This can be calculated by:

Rainfall weighted ci = $(Q_r c_r + Q_{iw} c_{iw}) / (Q_r + Q_{iw})$

Where

ci = concentration of water entering the soil (usually expressed as EC)
Q_r = quantity of rainfall
c_r = concentration of input water due to rainfall
Q_{iw} = quantity of irrigation water
c_{iw} = concentration of input water due to irrigation

This formula assumes that the salt content at the base of the root zone matches the concentration of water draining beneath the root zone. A range of factors influences the actual concentration (SalCon, 1997).

TABLE 47 – APPROXIMATE LEACHING FRACTIONS FOR DIFFERENT SOIL TEXTURES

Soil Texture	Assumed LF	Range
Sand	0.4	0.3-0.6

Loam	0.15*	0.1-0.3
Light clay	0.15	0.05-0.2
Heavy clay	0.1	0.05-0.3
Clay soils with heavy clay subsoils or very poor structure with poor subsoil wetting	0.05	0.002-0.1

Source: Yo and Shaw 1990 cited by SalCon (1997).
*SalCon (1997) cites this value as "1.15". It is assumed that 0.15 is correct since this fits within the range for a loam soil.

For permeable soils, salt leaching can be managed by varying water applications according to this principle. Irrigating more frequently reduces plant water stress, dilutes the soil solution and increases salt leaching providing the root zone is wet. In well-drained soils with a deep water table, regular irrigation during the growing season assists in salt leaching. For crops that vary in their salinity tolerance with growth stage, match the water quality to the growth stage. Alternatively, salt accumulations during the growing season should be irrigated specifically to leach salt. Ponding works efficiently for slowly draining soils, while leaching in small amounts is more efficient for all other soils. However, for slowly permeable soils (drainage of 1-10 mm/d), the soil properties and sodicity mainly control leaching rates, reducing the effectiveness of irrigation water management. Adding further water for leaching may increase soil salinity by adding salts at rates exceeding the removal rate via leaching. In this situation, rainfall is a more suitable source of water for leaching (SalCon, 1997). To maximise the leaching of salt with rainfall, the soil profile should be refilled with water to its upper drained limit before the autumn rain break (Smith *et al.,* 1996).

The irrigation method chosen affects salt accumulation. Well-designed flood irrigation systems spread water evenly. In cracking clay soils flood irrigation provides good soil water recharge and potential for leaching if the soil is cracked before irrigation. Furrow irrigation accumulates salt in adjacent rows through capillary rise and evaporation from the peaks of the rows. Planting in the furrow or on the side of the rows reduces the salt concentration. Sprinkle irrigation can cause leaf damage through salt deposition on leaves. Irrigating below the leaves or at night (which minimises the concentration of salt through evaporation) reduces this effect. The relatively low application rate of sprinkle irrigation can surface seal cracking clays reducing soil wetting. Trickle irrigation contributes to salt accumulation in the soil surface. When it rains, salts may leach through the root zone at concentrations that can kill vegetation. Applying mulch to minimise evaporation helps (SalCon, 1997).

When planning for salt removal by leaching, it is important to consider climatic variations between years. In dry years, there may be insufficient leaching to balance the salts added. Also, the effluent for irrigation may be more concentrated in these years because of reduced rainfall to the ponds and higher evaporation (Bond, 1998).

Soil salinity on effluent reuse areas can be managed through appropriate reuse practices and salt leaching. The expected increase in soil salinity from effluent reuse should be calculated prior to irrigation. The use of the DNRM (Queensland) SALF model for calculating predicted soil salinity after effluent irrigation is recommended since this is able to consider the effects of leaching.

8.3.7. The Fate of Leached Salts

The ultimate fate of leached salts is important. This depends upon the stratigraphy beneath the site and groundwater conditions. With porous soils, up to 6 t salt/ha/m depth may be stored between the root zone and the water table. Once it fills, salt will reach the water table. Some lateral movement is also likely (Bond, 1998).

The effect of salt moving to the groundwater depends on the pre-existing quality of the groundwater and the change in the salinity of the affected water. There is a school of thought that it is acceptable to add further salt to saline groundwater. An alternative view is that salt leaching should not change the beneficial use of the groundwater. The latter view sits better from a sustainability perspective. It is not acceptable to change the suitability of water quality for any purpose. Salt leaching to the aquifer will be diluted by the aquifer. The extent of dilution depends on the recharge rate, the groundwater flow rate and the size of the effluent irrigation area (a larger area will potentially have a greater effect on down-gradient groundwater quality because there will be less dilution (Bond, 1998)). Nevertheless, Bond (1998) indicates that monitoring of groundwater beneath effluent irrigation areas is an essential indicator of environmental performance.

8.4. Sodicity

This section includes a detailed study of sodicity and examines the effects of reusing the effluent and solid by-products of intensive livestock production on soil sodicity. It includes information on the following:

- Definition of sodicity
- Sodicity effects on the environment
- Measuring sodicity (SAR in effluent and solids)
- Analysis methods (standards, costs, accuracy, sample storage & handling)
- Sample collection, storage and handling
- Interpretation of sodicity results
- Managing sodicity
- Sodicity indicators of sustainability

8.4.1. Definition of Sodicity

Effluents and other liquids (including the soil solution) are sodic if there is a high exchangeable sodium percentage (ESP) relative to the total exchangeable cations. The Sodium Absorption Ratio (SAR) is used to evaluate sodicity in liquids.

Soil sodicity occurs when the ratio of exchangeable sodium ions to other exchangeable cations is sufficient to influence the swelling and dispersion behaviour of soils (Rengasamy and Churchman 1999). Australian soils are often regarded as being sodic if more than 5%-6% of the total cations present (or Cation Exchange Capacity (CEC)) is sodium, i.e. if the exchangeable sodium percentage (ESP) is more than 5% or 6%. However, this is a generalised view since not all soils with an ESP exceeding 6% show the swelling and dispersion behaviour associated with sodicity (Rengasamy and Churchman 1999). For a given soil ESP, sodicity is more likely to be exhibited if the soil also has a low ratio of exchangeable calcium to exchangeable magnesium. For a given soil ESP, sodicity is less likely to be exhibited in saline soils. There is no specific ESP at which soils become sodic.

There is no specific ESP at which soils become sodic. However, Australian soils are generally sodic if the ESP exceeds 6%.

A high exchangeable potassium percentage (e.g. >30%) in the soil may cause similar clay dispersion behaviour to sodic soils (e.g. Biswas and Higginson 1997). This is relevant for some effluent reuse areas.

8.4.2. Sodicity Effects on the Environment

Australia has the largest area of sodic soils of any continent. Some soils are naturally sodic because of the nature of their parent materials or because of the past presence of shallow sodic water tables in low permeability soils (Shaw 1999). Induced sodicity in other soils is a side effect of rising water tables from land clearing and irrigation with sodic water (Rengasamy and Churchman 1999).

The causes and effects of sodicity can be understood through a basic knowledge of soil chemistry. In the soil, ion substitution in the clay mineral lattice creates a negative charge. This attracts exchangeable cations. The composition of the adsorbed cation layer depends on the type of clay and the constituents and concentration of the surrounding soil solution. When there is a high sodium concentration in the soil solution, the concentration of sodium bound to the clay particles will also be relatively high. The attractive forces between clay particles are strongest when calcium is the dominant cation on the clay surface and are lowest when sodium dominates. This occurs because the cation layer for the calcium cations is smaller, reducing the distance between clay particles. When the soil is wetted, repulsive forces associated with hydration cause soil swelling. Swelling is lower if the cation layer is thin (e.g. calcium) and greater if the cation layer is thick (e.g. sodium). Consequently, when the proportion of exchangeable sodium attached to clay particles is high, there is increased particle separation and dispersion on wetting (Rengasamy and Churchman 1999, SalCon 1997). Irrigation with sodic (high SAR) water or effluent can induce soil sodicity by replacing other cations on the clay mineral exchange sites with sodium (SalCon 1997). However, this does not necessarily occur if the effluent or the soil being irrigated is also saline since salinity inhibits the clay swelling and dispersion.

An ESP as low as ~6% can cause soil dispersion. The dispersion and movement of small clay particles through the soil can have a number of effects:

1. Pore spaces within the soil may be blocked, restricting the movement of water and air. This may cause waterlogging. It may also reduce the plant available water capacity.

2. Reduced water leaching causes salt accumulation and the development of saline subsoils.

3. A hard-setting or massive surface or subsoil that further impedes water entry and storage may form. This inhibits plant germination and may restrict root development.

4. Because of reduced natural aggregation, structure is cloddy or absent when soils dry making sodic soils erosion prone (SalCon 1997, Rengasamy and Bourne 1997).

Problems such as hard-setting, waterlogging, poor infiltration and poor root development arise mainly because of subsoil swelling and dispersion (Rengasamy and Churchman 1999). Exchangeable sodium levels frequently increase with depth in soil profiles. However, since salinity levels also tend to increase with depth, soil dispersion and swelling are often suppressed (Rengasamy and Churchman 1999). This is because salt tends to flocculate the soil, which acts to counter soil dispersion from sodicity (SalCon 1997).

8.4.3. Measuring Sodicity

The sodium absorption ratio (SAR) measures sodicity in liquids since there is a close relationship between SAR and soil ESP. SAR is the amount of sodium relative to calcium and magnesium in a soil solution or water that approximates the exchangeable sodium percentage (ESP) of the soil (SalCon 1997).

$SAR = [Na]/(0.5 * [Ca]+[Mg])^{0.5}$
Where concentrations are in meq/L (SalCon 1997).

However, this is an imperfect measure since it is the concentration of cations available for adsorption rather that the absolute cation concentration that determines the subsequent ESP of irrigated soils (Halliwell *et al.*2001).

Soil ESP can be calculated from SAR using the relationship of USSL (1954) (cited by SalCon 1997):

ESP = (100 (-0.0126 + 0.01475 SAR)) / (1 + (-0.0.0126 + 0.01475 SAR))

Because divalent cations are preferentially adsorbed onto clay exchange sites, the proportions of Ca^{2+}, Mg^{2+} and Na^{+} on the soil exchange do not match the proportions in the soil solution. The reverse equation can be used to derive SAR from ESP:

$SAR = 0.6906\ ESP^{1.128}$
(This equation is applicable for ESP values of 0-50). (SalCon 1997).

Because of the relationship between sodicity and salinity, it is also important to measure the EC of the effluent.

Residual alkali (RA) provides a measure of the effect of effluent irrigation on soil properties. RA measured the excess of sodium bicarbonate and carbonate ions in the water over calcium and magnesium ions. When these salts combine with calcium and magnesium in the soil solution, they are removed by precipitation leaving an excess of sodium ions in the soil. However, RA on its own in not a useful measure of sodicity hazard since water may have a high SAR but a low RA (SalCon 1997).

Measuring the SAR and EC of effluent provides a good guide to the sodicity potential of the effluent for irrigation.

An indicator of soil sodicity is the Exchangeable Sodium Percentage (ESP), which is the amount of sodium ions adsorbed by clay particles as a percentage of total Cation Exchange Capacity (CEC) (SalCon 1997). The calculation for exchangeable sodium percentage (ESP) is:

ESP (%) = (Exchangeable Sodium (meq/100 g)/CEC (meq/100 g)) * 100

Cation Exchange Capacity (CEC) is the total amount of cations on the surface layer of clay materials that are readily exchanged with other cations available in solution, expressed as millequivalents per 100 grams of dry clay (meq./100 g) (SalCon 1997). The dominant exchangeable cations in most soils are: Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}. Exchangeable cations present in smaller percentages can include: NH_4^{+}, Cu_2^{+}, Co_2^{+} and Zn_2^{+}. Aluminium, iron and hydrogen cations may also be present in acidic soils (Rengasamy and Churchman 1999).

CEC (meq./100 g) = Exch. Ca + exch. Mg + exch. Na + exch. Ca

where all units are in meq./100 g (Rengasamy and Churchman 1999).

The calcium to magnesium ratio and the exchangeable potassium percentage also play a role in soil dispersion. Consequently, it is useful to find the ratio of exchangeable calcium (meq/100 g) to exchangeable magnesium (meq/100 g). A ratio of less than 2 is likely to be associated with soil structural problems in some soils. It is also useful to measure the exchangeable potassium level and then find the exchangeable potassium percentage.

EKP (%) = (Exchangeable Potassium (meq/100 g)/CEC (meq/100 g)) * 100

Soil SAR can be measured. However, relationships between ESP and SAR in soil solutions depend on both the method of extraction and soil properties. Hence, these should be applied with caution (Rengasamy and Churchman 1999).

The concentration of salts in solution is a particularly important determinant of the expression of sodicity for a given soil (Rengasamy and Churchman 1999). Consequently, examining ESP alone may not be the most accurate indicator of sodicity. Also, finding the calcium to magnesium ratio and EKP provides useful information. Simple field tests are available to assess the degree of turbidity produced when soil is gently mixed with distilled water. Sodic soils disperse, creating turbidity, whereas non-sodic soils do not (Rengasamy and Bourne, 1997).

Soil dispersion can be directly measured using simple field tests. However, ESP, calcium to magnesium ratio and EKP are all useful indicators of the risk of soil dispersion. In this instance, ESP is the preferred measure of sustainability since it provides an objective and widely accepted measure.

8.4.4. Analysis Methods

To calculate ESP, it is necessary to determine the cation exchange capacity (CEC) and exchangeable sodium. A study by Maheswaran and Peverill (1995) (cited by Rengasamy and Chuchman (1999)) revealed large variations in analysis results for exchangeable sodium and CEC measurements made on the same samples by different Australian laboratories.

CEC and exchangeable sodium should be determined using the methods in Rayment and Higginson (1992). Where possible the same laboratory and analysis method should be used for samples collected regularly from monitoring sites.

See Appendix D for further details.

8.4.5. Sample Collection, Storage and Handling

See Appendix F for details.

8.4.6. Interpretation of Sodicity Results

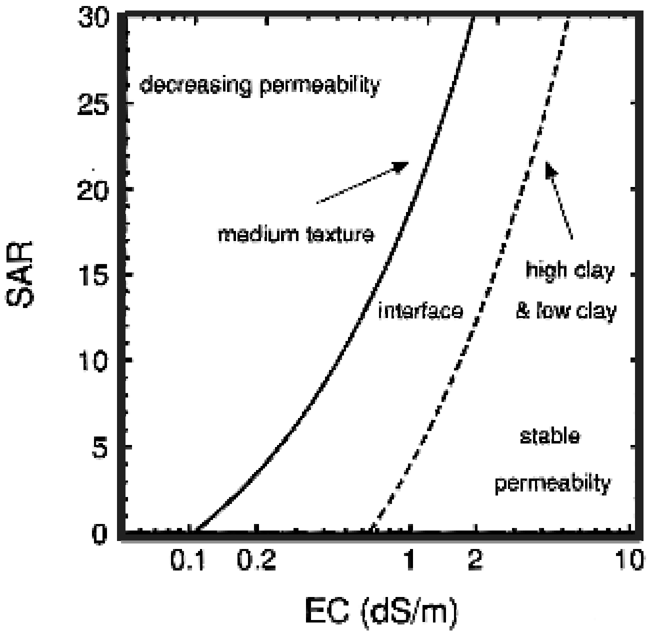
Gill (1984) provided sodicity classes for irrigation waters based on SAR. These are presented in Table 48. However, soil responses to SAR depend on salinity also. As the salinity of the effluent increases, clay swelling and dispersion are inhibited by the increasing electrolyte effect of the soil solution. Consequently, the acceptable SAR increases with effluent salinity. This relationship for various soil types is shown in Figure 5. SAR-EC combinations to the left of the line have unstable soil profile permeability. Medium textured soils are more stable than either high or low clay soils (Shaw *et al.* 1994).

TABLE 48 - SODICITY CLASSES FOR IRRIGATION WATERS

SAR	Sodicity Class
<3	1 No sodium problem
3-6	2 Low sodium, few problems except with sodium sensitive crops
6-8	3 Medium sodium, increasing problems, use sodium and not sodium sensitive crops
8-14	4 High sodium, not generally recommended
>14	5 Not suitable for irrigation

(Gill 1984).

FIGURE 5 – RELATIONSHIP BETWEEN SODIUM ABSORPTION RATIO (SAR) AND EC FOR IRRIGATION WATERS THAT DEFINE THE INTERFACE BETWEEN THREE GENERAL SOIL TEXTURES



(Reproduced from Shaw *et al.* 1994).

Salt Action (1999) state that water is sodic if the SAR exceeds 3.

Northcote and Skene (1972) developed sodicity classification criteria for assessing Australian soils (Northcote and Skene (1972) cited by SalCon (1987)). These are presented in Table 49. These data should be interpreted in relation to soil properties since the influence of sodicity on soil behaviour depends on the soil properties (Shaw 1999). For instance, sandy soils have higher ESP thresholds for sodicity than clay soils. For unprotected surface soils that are subject to water erosion, an ESP value of 3 may be a better indicator of non-sodic soils. However, a subsoil ESP of 15 or more may be acceptable, particularly for cracking clays (SalCon 1997).

TABLE 49 – CRITERIA FOR ASSESSING SODICITY IN SOILS

Criteria	Description
ESP <6	Non-sodic
ESP 6-14	Sodic
ESP >15	Strongly sodic

(Northcote and Skene (1972) cited by SalCon (1987)).

While ESP provides a guide for sodicity assessment, it is important to consider the behaviour of the soil also. A soil ESP determination of 6 or more should trigger further investigation. This investigation would take the form of comparison with analysis results for background plot soil samples. An ESP exceeding 50% of background level in any soil layer would be considered unsustainable.

8.4.7. Managing Sodicity

Several management practices are available to manage or ameliorate sodicity.

1. Gypsum

Gypsum is the most common amendment for sodicity, but is only effective in non-saline soils. Addition of gypsum to sodic, non-saline soils causes calcium displacement of some of the sodium ions from the clay particles. In the short-term, clay swelling and dispersion decline as the gypsum dissolves in the soil water creating a salt solution. In the longer term, calcium cations from the gypsum displace some of the sodium cations that are attached to the clay. Calcium clays are less prone to swelling and dispersion than sodic clays. The displaced sodium then leaches below the plant root zone (Lines-Kelly, 2000).

Although field and laboratory methods can identify the occurrence of sodicity, there is no accurate method to determine appropriate gypsum application rates for sodicity amelioration. This is often controlled by cost (Rengasamy and Churchman 1999).

Rengasamy and Bourne (1997) suggest the following rates:

- 2.5 t/ha gypsum for sodic soils.
- 5 t/ha for sodic alkaline soils or highly sodic soils. Higher rates are needed since gypsum is less effective under alkaline conditions.
- 10 t/ha gypsum for highly sodic soils under irrigation.
- If using lower quality gypsum, increased application rates will be needed.
- For alkaline soils, consider planting acidifying legumes to lower the pH

Moody (2000) suggests the quantity of gypsum needed to amend sodic soils can be crudely estimated by calculating the quantity of exchangeable sodium needing displacement from exchange sites for a given soil depth to lower the ESP to an acceptable level. However, since it is unlikely that it will be economically feasible to apply this quantity of gypsum in one application, rates of 2.5-5 t/ha are suggested (Moody 2000).

Gypsum quality is assessed by purity and fineness. In NSW, gypsum purity is defined by the sulfur percentage (w.b.). Most of the gypsum sold in NSW is calcium sulfate dihydrate

($CaSO_4.2H_20$). This contains 18.6% sulfur if pure. Fine gypsum is superior to lumpy gypsum, which dissolves only slowly. The ease of spreading should also be considered when selecting gypsum (Abbott and McKenzie, 1996).

Gypsum is best applied before the seasonal break since rainfall dilutes surface soil salinity, promoting soil dispersion (Garner *et al.,* 1995).

2. Lime

For acidic, sodic soils cultivating in lime can improve soil structure while also raising the pH (Lines-Kelly, 2000). Aim for a soil pH (water) of 6 or higher. To raise soil pH by about 1 pH unit in the top 10 cm add:

- Sands - 1-2 t/ha of lime (lime rates exceeding 2.5 t/ha on light sandy soil may induce manganese deficiency).
- Loams - 2-3 t/ha of lime.
- Clays - 3-4 t/ha of lime (Rengasamy and Bourne 1997).

Because lime is much less soluble than gypsum, it is generally much slower in being effective (Rengasamy and Bourne 1997). Also, it is unsuitable for use with alkaline soils (Rengasamy and Churchman 1999).

3. Apply Sulfur or Sulfuric Acid

Adding sulfur to sodic soils helps if there is a high concentration of calcium carbonate at the sodic depth. Adding sulfuric acid reduces alkalinity but will only reduce sodicity if there are low calcium or magnesium levels (SalCon 1997).

4. Deep Ripping

Deep ripping may help to improve soil structure (Rengasamy and Churchman 1999).

5. Adding Organic Matter

Building up soil organic matter levels, for example through addition of manure solids, helps to improve soil structure (Rengasamy and Bourne 1997).

6. Irrigating with Effluent With a Low SAR

Shaw and Thorburn (1985a cited by SalCon 1997) developed a relationship between soil EC_{se} and ESP. This relationship can be used to develop guidelines for a permissible SAR of effluent for irrigation for soils with different textures. This SAR should maintain soil stability under the high leaching situations resulting from heavy rainfall.

TABLE 50 – PERMISSIBLE SAR OF IRRIGATION WATER TO MAINTAIN STABLE SOIL FOLLOWING HEAVY RAINFALL

Clay Content (%)	Soil Texture	Permissible Irrigation Water SAR* Clay mineralogy expressed as $mole_c$/kg				
		<0.35 non-cracking**	0.35-0.55 non-cracking	0.55-0.75 cracking**	0.75-0.95 strongly cracking	>0.95 very strongly cracking
<15	Sand, sandy loam	>20	>20	>20	>20	>20
15-24	Loam, silty loam	20	11	10	10	8
25-34	Clay loam	13	11	8	5	6
35-44	Light clay	11	8	5	5	5
45-54	Medium clay	10	5	5	5	5
55-64	Medium-heavy clay	5	5	5	4	4
65-74	Heavy clay	-	4	4	4	4
75-84	Heavy clay	-	-	4	5	5

* Values calculating assuming surface soil EC equal to undisturbed soil in Lockyer Valley, modified from Shaw and Thorburn (1985a) at 2000 mm rainfall
** Cracking or non-cracking only applicable if clay content exceeds about 35%

7. Irrigate for Longer Time Periods

Since sodic soils have lower infiltration rates, longer irrigation periods are needed to wet the soil (SalCon 1997).

8. Continue Irrigating with Undiluted Effluent

Shandying effluent with higher quality water can reduce the salinity hazard. However, this creates problems if applied to sodic soils. Because of the interaction between sodicity and salinity in the soil, as long as effluent irrigation continues, soil structure is unlikely to deteriorate since the high salinity of the effluent counterbalances the high SAR and resulting ESP of the soil. However, this can change when effluent irrigation stops (Bond, 1998). It is also particularly evident when rain follows irrigation with sodic water. In this situation, the total salt content of the soil solution falls due to leaching. However, the ESP is reduced by a lesser amount since the number of exchangeable ions (those held by soil) in a set volume of soil is usually 50-500 times greater than the number of ions in the soil solution. As a result, there is a much smaller number of calcium and magnesium ions in the soil solution to replace the exchangeable sodium. If there is insufficient salt to offset the effects of exchangeable sodium, clay swelling and dispersion will occur causing a reduction in soil permeability and infiltration rates (SalCon 1997).

8.4.8. Indicators of Sustainability for Salinity and Sodicity

Sustainability issues for managing salt and sodium in effluent reuse include:

1. Preventing salt accumulation in the root zone.
2. Maintaining soil structure.
3. Avoiding reductions in the productivity of plants grown on reuse areas due to excess salt.
4. Minimising off-site effects, particularly increases in the salt content of groundwater.

In evaluating the load based licensing protocols for salinity, Hird (1998) argues that the water supply for any activity has a salt loading prior to use and that it is inappropriate to incorporate the salt entering from this source in the load based licensing fee structure. Nevertheless, the total salt load of the effluent needs to be considered when managing effluent irrigations to prevent soil degradation and reductions in plant yields.

A long-term objective for any reuse area should be to ensure that there are no consistent increases in soil salinity. Clearly there may be pronounced increases in soil salinity through the addition of effluent or solid by-products, particularly in the topsoil layer. However, these increases need to be offset by leaching losses to ensure no consistent and signficant increases in soil salinity in the subsoil layers. In dry years in particular, leaching rates will be lower and it will take longer for salt removal to occur. Based on the data in Table 45, soils with an EC_{se} of up to 1.9 dS/m fall into the very low to low salinity rating. Thereafter, any increase in EC_{se} of 2.5 dS/m would shift the soil salinity rating by less than one salinity class, the EC_{se} ranges for different classes being:

- <0.95 dS/m for a "very low" salinity rating.
- 0.95-1.9 dS/m (or a range of 0.95 dS/m) for a "low" salinity rating.
- 1.9-4.5 dS/m (or a range of 2.6 dS/m) for a "medium" salinity rating.
- 4.5-7.7 dS/m (or a range of 3.2 dS/m) for a "high" salinity rating.
- 7.7-12.2 dS/m (or a range of 4.5 dS/m) for a "very high" salinity rating.
- >12.2 dS/m for an "extreme" salinity rating.

Consequently, it is considered that triggers for further investigation should be any EC_{se} increase of 2.5 dS/m compared with similar soil sampled from 'virgin' sites <u>and</u> any result that places the salinity rating at "medium" or above. Soil EC_{se} should be determined at 50-60 cm (or base of root zone).

It is suggested that soil sampling should occur at the end of the main growing season when the plants grown on the area have had time to assimilate nutrients and salts have had time to leach through the soil profiles. It is suggested that EC_{se} at the base of the root zone would act as a sustainability indicator, but surface and upper subsoil levels should also be monitored for agronomic purposes and to monitor salt movements through the soil profile.

The primary sustainability indicator for sodicity is ESP measured at depths of 0-10 cm and 50-60 cm (or base of root zone). A trigger for further investigation is a soil ESP exceeding 6%. If the ESP exceeds 6%, comparison with the soils of a background plot is necessary. An ESP increase exceeding 150% of background levels (e.g. from 4% to more than 6%) in any soil layer is considered unacceptable.

8.5. Summary of Sustainability Indicators

This section includes the sustainability indicators that have been judged to provide the best practical and objective measure of sustainability. It is expected that in the vast majority of cases they will provide a good tool for the assessment of sustainability. However, it is important to recognise that non-compliance with the triggers associated with the indicators does not necessarily suggest that a system is unsustainable. In these instances, operators of piggeries and cattle feedlots should be able to use other indicators to demonstrate sustainability.

The appropriateness of the indicators has been determined using an evaluation of the criteria of Walker and Reuter (1996).

Able to be measured easily and economically - The indicators provided are based on determining the sustainability of a reuse practice by firstly measuring soil parameters that are used for agronomic purposes and are thus well-known and easily determined.

Fewer rather than many indicators - The number of indicators have been kept to a minimum by recommending single parameters that determine sustainability. For example, to determine if there is any leaching of nutrients down the profile the most mobile parameter (nitrate-nitrogen) is recommended, not a suite of elements.

Able to be measured at achievable and appropriate levels of precision – Well-known and proven tests have been selected. This should ensure all accredited laboratories that are able to perform the test are able to follow a prescribed method. Significant information is also available in the appendices for handling samples appropriately.

Simply quantified – Simply quantifiable triggers of sustainability have been determined that are to be used with further checks, such as levels compared to background data and the risk of contamination e.g. groundwater.

Interpretable – By quantifying triggers of indicators, sample data can be compared and interpreted.

Able to indicate spatial and temporal variation – Spatial variation is accounted for by ensuring separate monitoring plots are set-up for different soil types, different land uses (crop type) and application type (liquid effluent or solid manure). These monitoring plots need to be representative of the re-use area. It is suggested that monitoring of soils be undertaken annually to investigate trends and averages over-time. More frequent monitoring would detect seasonal variations, but these would not be useful in determining long-term sustainability.

Able to suit all levels of enterprises – By keeping the number of parameters small and well defined they are capable of being used in any size enterprise.

Ease of capture – This is related to the age of a method. As methods are more highly developed they become easier to perform. Phosphorus methods are still being refined and as such are not easy to quantify, however the most appropriate methods currently available have been chosen.

Total cost /ha/test – The most economical method has been chosen, by ensuring the minimum number of parameters to determine sustainability.

Existence of a standard method of estimation – Only standard methods have been suggested, but where promising research has shown highlighted improved methods (e.g Varying P sorption based on soil type and climate, not standard 0.5 mg/L threshold – Redding pers comms. (2002)), they have been included in recommendations.

Interpretation of criteria available – Expected and threshold values have been determined for each sustainability indicator.

Significant on a property scale – The indicators presented are used because they are meaningful on a farm scale and are directly related to what is happening on the farm, with the aim of protecting off-site resources (e.g. ground and surface waters).

Low error associated with measurement – Provided collection methods are followed as per the appendices, errors will be minimised. The use of long-term trends and averages also over-comes possible errors.

Known response to land management or disturbances – Changes in the levels of the sustainability indicators can be directly linked to land management.

Trend indicators are mappable – It is suggested that monitoring be checked against background in order to establish trends.

Generic rather than diagnostic – The indicators developed are generic and can be used for most industries producing organic by-products that are reused on land for cropping.

8.5.1. Nitrogen

Nitrogen in the nitrate form is extremely mobile and readily leached. Consequently, high nitrate-nitrogen levels in the subsoil pose a risk to groundwater. Once the nitrogen moves below the plant root zone, it is no longer available for plant uptake and can leach to groundwater. An obvious sustainability indicator of nitrogen in reuse areas is the nitrate-nitrogen concentration below the plant active root zone.

Subsoil nitrate-nitrogen concentrations exceeding a soil solution concentration of 10 mg nitrate-N/L may produce some nitrogen leaching losses. The 10 mg/L nitrate-N is based on the Australian Water Quality Guidelines for Fresh and Marine Waters (ANZECC, 1992) which state that nitrate-nitrogen concentrations should not exceed the 10 mg/L level in groundwater used for human consumption.

Applying a drinking water quality standard is likely to be overly stringent in many cases since the groundwater under reuse sites is unlikely to be used for human drinking water and it assumes there is no further losses or dilution before it reaches the groundwater. This limit is commonly exceeded in normal agricultural soils. Vertosols, for example, can have relatively high nitrate-nitrogen levels in their natural state. When assessing the sustainability of a reuse practice in terms of nitrogen levels, a number of factors need consideration, including the value or use of surrounding groundwater resources (human consumption, animal consumption, irrigation etc), the depth to groundwater, soil type overlaying the groundwater (e.g. clay) and baseline levels of nitrate-nitrogen in soil below the active root zone.

Consequently, a nitrate-nitrogen limit of 10 mg/L below the active root zone is suggested only as a trigger for further investigation. This further investigation would

involve the comparison of monitoring results from the reuse area with those of the same soil that has not had effluent or manure applied (e.g. under a fenceline). If the level of nitrate below the active root zone show signs of build-up over-time (nitrate bulges), the reuse practices employed will need review in line with the forward management plan of the operation. Thus comparing nitrate-nitrogen monitoring results against baseline data provides a measure of the nitrogen sustainability of a reuse area.

Other matters to consider when determining the sustainability of the reuse practice in terms of nitrogen include the risk of nitrate moving off-site in surface water and groundwater, the quality (value) of the groundwater and the amount of deep drainage of the soil of the reuse area. These need to be evaluated as part of the risk assessment of the reuse area.

The amount of deep drainage will vary with soil type, rainfall, the amount of effluent or fresh water irrigated and the type of crop production. For example, deep drainage may range from 10mm/yr to 150 mm/yr for a black vertosol and a loamy-sand respectively, when a crop of improved pasture is grown and a total of 750 mm of rainfall and effluent irrigation is applied. With 10 mg/L of nitrate-N in the deep drainage, this represents a loss of 1 kg of N/ha/yr for the black vertosol and 15 kg of N/ha/yr for the loamy sand.

The depth of the root zone depends on the crop type, soil depth, climatic condition and whether the crop is irrigated. In some cases the active root zone depth may be 1.5 – 2.0 m and even deeper (e.g. dryland lucerne). Thus, sampling below the root zone may not always be practically and economically feasible. Sampling to a depth of at least 60 cm is recommended, although deeper sampling (to the base of the root zone) may be required if there are concerns about nitrate leaching.

8.5.2. Phosphorus

Skerman (2000) states that significant leaching of phosphorus generally occurs only when the soil is heavily overloaded with phosphorus. Table 36 gives surface soil available phosphorus concentrations that will meet plant requirements and should not result in significant losses to surface water, provided runoff is controlled via good design and management. Since these limits are commonly exceeded in normal agricultural soils, they are triggers for further investigation via comparison against results from 'virgin' soils receiving no effluent or manure or if there are doubts about the sustainability of the reuse practice.

The Department of Land and Water Conservation (NSW), Soil And Land Information System (SALIS) database ranks various chemical test results for NSW soil tests, including Bray P. These rankings are shown in Table 37. The high ranking of 20-25 mg/kg Bray P in the surface soil could be used as a guideline measure of a trigger for further investigation. This further investigation could include comparison against background data.

Redding pers. comms. (2002) developed limits of available phosphorus in the surface soil for the BSES method, based on the same principles as the limits for Colwell (mean + one standard deviation) depending on the level of clay. These are shown in Table 38. For soils with less than 30% clay the guideline level is 31 mg/kg of BSES P. For soils with greater than 30% clay the guideline level is 131 mg/kg of BSES P. It should be noted that these numbers are derived from a relatively small data-set and may need to be refined when more data is available.

Both the Bray and BSES may be more appropriate measures of available P in certain soils (e.g. acid).

To investigate any possibility of P leaching, particularly with sandy soils, measurement of available P levels at 50 – 60 cm (or the base of the root zone) is also suggested.

The soil profile to the base of the crop root zone should be considered the safe storage interval for applied phosphorus. To prevent excessive leaching of phosphorus below the root zone it is recommended that the equilibrium solution concentration of phosphorus of 0.5 mg P/L be used to estimate the safe phosphorus storage capacity. Thus, phosphorus applications exceeding removal by the plant material should not go beyond the phosphorus sorption capacity of the soil at an equilibrium solution concentration of phosphorus of 0.5 mg P/L. However, this soil solution concentration level needs review pending the findings of the recent Redding work. It would be possible to generate appropriate soil solution concentration levels for different soil types and regions from currently available data.

A reuse area should be used to store phosphorus only if it good cropping land and providing a plan is in place to continually crop the area after effluent or solids reuse has ceased to remove the stored phosphorus as it is released. The phosphorus storage capacity of the reuse area should also be determined by measuring a P sorption isotherm every five years.

The P sorption capacity of the soil will generally change down the soil profile due to decreasing levels of available P and changes in soil texture. Phosphorus sorption capacity can be determined by a single average test of the soil profile to the base of the root zone to reduce significant analysis costs. However, it may be beneficial for producers to test the P sorption capacity of different soil layers in some instances.

8.5.3. Salt

A long-term objective for any reuse area should be to ensure that there are no consistent increases in soil salinity. Clearly there may be pronounced increases in soil salinity through the addition of effluent or solid by-products, particularly in the topsoil layer. However, these increases need to be offset by leaching losses to ensure no consistent and signficant increases in soil salinity in the subsoil layers. In dry years in particular, leaching rates will be lower and it will take longer for salt removal to occur. Soils with an EC_{se} of up to 1.9 dS/m fall into the very low to low salinity rating. Thereafter, any increase in EC_{se} of 2.5 dS/m would shift the soil salinity rating by less than one salinity class. Consequently, it is considered that a trigger for further investigation should be any EC_{se} increase of 2.5 dS/m compared with similar soil sampled from 'virgin' sites and any result that places the salinity rating at "medium" or higher. Soil EC_{se} should be determined at a depth of 50-60 cm (or base of root zone).

It is suggested that soil sampling should occur at the end of the main growing season when the plants grown on the area have had time to assimilate nutrients and salts have had time to leach through the soil profiles. It is suggested that EC_{se} at the base of the root zone would act as a sustainability indicator, but surface and upper subsoil levels should also be monitored for agronomic purposes and to monitor salt movements through the soil profile.

If further investigations are warranted, the soil Na^+ + Cl^- concentration throughout the profile should be determined for the reuse and background sites since sodium chloride is the main salt of interest from a soil degradation perspective. The soil Na^+ + Cl^- concentration of the soil should be less than 150% of background levels.

8.5.4. Sodicity

Sodicity is important in effluent reuse schemes because of the relatively high sodium content of the effluent and the adverse effects of sodicity on soil structure.

The primary sustainability indicator for sodicity is ESP measured at depths of 0-10 cm and 50-60 cm (or base of root zone). A trigger for further investigation is a soil ESP exceeding 6%. If the ESP exceeds 6%, comparison with the soils of a background plot is necessary. An ESP level exceeding 150% of background (e.g. from 6% to more than 9%) in any soil layer is considered unsustainable. It is acknowledged that soil with an ESP exceeding 6% is not necessarily dispersive, particularly if saline. However, non-dispersive saline soils with a high ESP have potential to become dispersive if the soil salinity declines in the future. For example, in high rainfall years, salinity may fall more rapidly than sodicity through increased drainage of the more soluble salts. Declines in soil salinity through drainage may also be more rapid than falls in sodicity after cessation of effluent reuse. Both these scenarios can give rise to soil dispersion. Consequently, calcium application is recommended where the soil ESP exceeds 6% and strongly recommended where it exceeds 9%.

Applying calcium to the soil in the form of high quality gypsum helps to displace sodium ions from the clay particles, making them available for leaching below the root zone. Consequently, an ESP level of 6% warrants gypsum application to amend the sodium imbalance while this is strongly recommended where the ESP has risen to 9%. For neutral to acidic sodic soils (ESP = 6-15%), apply 2.5 t/ha gypsum. Gypsum is less effective for alkaline soils, so a gypsum application rate of 5 t/ha is recommended for sodic alkaline soils. For highly sodic soils (ESP exceeding 15%), apply gypsum at 5 t/ha. For highly sodic, alkaline soils, consider planting acidifying legumes. If highly sodic alkaline soils are fully irrigated, gypsum application rates of up to 10 t/ha may be more appropriate (Rengasamy and Bourne, 1997).

8.5.5. Soil pH

Soil pH is important since it influences the availability of some nutrients. The pH throughout the profile should be within the range of 5-8. This has implications for nutrient uptake by plant growth since it may inhibit the availability of desirable nutrients or increase the availability of toxic elements.

9. Practicalities and Realities of Effluent and Solid By-Product Reuse

9.1. Introduction

Most regulatory agencies take a very long term and conservative view regarding effluent and solid by-product reuse. This effectively eliminates response to the short-term variations that exist in nature (e.g. climate). Regulatory requirements specify that the same amount of nutrients should be applied uniformly over the whole area each year. On the other hand, some farmers believe that heavy applications of manure (once off) are more practical and do not cause adverse environmental impact. However short-term, heavy applications to ensure availability of nutrients for crop growth are acceptable if the area is managed sustainably in the long-term.

This section aims to address the issue of nutrient availability, response to climate, short-term heavy applications and spelling/rotating areas from a practical and an environmental viewpoint.

To maximise crop yields and nutrient removal from a site, the crop needs to be supplied with adequate nutrients and water to maximise growth. Piggery and cattle feedlot effluents and manures are not balanced fertilisers (i.e. nutrient content of the manure or effluent are not the same ratio as the nutrient requirement of the crop). Solid by-products such as piggery sludge and feedlot manure typically have nitrogen to phosphorus ratios of 1:1 to 2:1, whereas most crops require nitrogen to phosphorus ratios of 5:1 to 10:1. So supplying sufficient solid by-product to meet a crop’s nitrogen requirements will supply an excess of total phosphorus. As a large percentage of the phosphorus is not immediately available to a plant and soils have varying capacities to store phosphorus, applications above plant use should be sustainable. The question is, how much can be applied for a reuse area to still be classed as ‘sustainable’?

Effluent and solid by-products from piggeries and cattle feedlots act as slow release fertilisers, since not all the nutrients are available to the plant upon or soon after application and the release of nutrients from their organic state is a complex and dynamic process. This is covered in more detail in Sections 8.1 and 8.2 of this document. This slow release of nutrients means that the crop has access to nutrients throughout the entire growth cycle. This is an environmental advantage over inorganic fertilisers that are available upon application or soon after application.

Work reported by Blair and EA Systems (2002) for the Cattle and Beef CRC at the Tullimba feedlot showed that the application of large manure tonnages (60 t/ha) every 3-4 years compared to annual application of 20-25 t/ha has some advantages. It reduces cultivations, thus decreasing the amount of deep and shallow soil compaction from manure spreading operations and also disturbance of soil structure. By balancing nutrients in manure with inorganic fertilisers, crop growth and nutrient uptake can be maximised, reducing pollution potential. Applying manure annually allows plants to take up the rapid flush of readily mineralisable nutrients from manure but not the residual nutrients released over time.

Monetary savings achieved by applying manure once every three to four years come through reduced cultivation frequency and decreased labour associated with manure application and cultivation operations. Consequently, applying inorganic fertiliser between manure applications made every 3 to 4 years compared to spreading manure annually provides both environmental and economic benefits.

Application of 60 t/ha of manure every three years produced higher yields and greater nutrient recovery than annual applications. Manure application enhanced the water holding capacity of the soil, reducing leaching and runoff.

9.2. Measured Case Studies

This section uses both case studies and research work to link theoretical calculations with reality. It includes examples of adverse environmental impacts from inappropriate reuse of intensive livestock effluent and solid by-products. These include problems such as soil acidification, soil structural problems (sodicity), groundwater contamination and surface water eutrophication. Also included are examples of long-term sustained effluent applications that have not caused adverse environmental impacts. This section also examines some theoretical research work showing the contribution of nutrient export from different land use practices, including piggeries and feedlots.

9.2.1. Two Case Studies of Long-Term Piggery Effluent Phosphorus Application

Redding (2001) reported on the long term land application of phosphorus in piggery effluent at two sites. Site P1 was a sodosol and had received 3700 kg of effluent P/ha over 19 years (averaging 195 kg of P/ha/yr). Site P2 was a dermosol and received a net load of 310,000 kg of P/ha over 30 years (averaging 10,300 kg P/ha/yr). At both sites surface bicarbonate and dilute $CaCl_2$-extractable molybdate-reactive phosphorus were significantly elevated. Phosphorus (as bicarbonate P) enrichment to 1.5 m was detected at P2. Elevated concentrations of $CaCl_2$-extractabe organic phosphorus (P_{OC}) were observed from the soil surface of P1 to a depth of 0.4 m. Even with the large applications of phosphorus, only site P1 displayed evidence of significant accumulation of P_{OC}.

This study showed that the increase in surface soil total phosphorus due to effluent irrigation was much greater than laboratory phosphorus sorption (>25 times for P1; >57 times for P2) for a comparable range of final solution concentrations. Precipitation of sparingly soluble phosphorus phases was evidenced in the soils of the P2 effluent application area. The author concludes that the relationship between laboratory sorption isotherm data and long-term sorption in effluent systems requires further investigation.

9.2.2. Eleven Piggery Effluent Reuse Sites

Redding *et al.* (2002) summarised phosphorus related properties of soils from 0-0.05 m depth at 11 piggery effluent reuse sites to explore the impact of effluent reuse on the potential for runoff transport of phosphorus. The sites investigated included several different soil types that had received effluent for 1.5 to 30 years. Effluent phosphorus application ratio ranged from 100 – 310 000 kg P/ha in total. Total phosphorus (P_T), bicarbonate extractable phosphorus (P_B) and soluble phosphorus were determined for the soils on both reuse and control sites for each application area. P_B increased between 1.7 and 15 times on the effluent reuse sites, compared to background areas at 10 of the 11 sites. Increase in P_B was strongly related to net phosphorus applications. Effluent application tended to increase the proportion of soil P_T in dilute CaCl2-extractable forms. The study concluded that current effluent management at many of the piggeries had failed to maximise the potential for phosphorus recapture. Ten of the case study effluent application areas received effluent-phosphorus at rates exceeding crop uptake. While this may not represent a significant risk of leaching where sorption retains phosphorus, it increases the risk of phosphorus transport by runoff.

9.3. Predicting Nutrient Export from Different Land Uses

Prentice and Walker (2002) used the Catchment Management Support System (WinCMSS) to predict the export of nutrients (nitrogen and phosphorus) and total suspended solids from different land uses in the Condamine Balonne catchment. Potential pollution sources, both point and diffuse, were inputs into the model and generation rates of the nutrient exported by land use were determined. Piggeries (164) and feedlots (103) were included in the modelling.

The results of the modelling predicted piggeries and feedlots contributed a very small percentage of the phosphorus export in the catchment (Figure 6). They also reported these percentages (Figure 6) were similar for nitrogen and suspended solids.

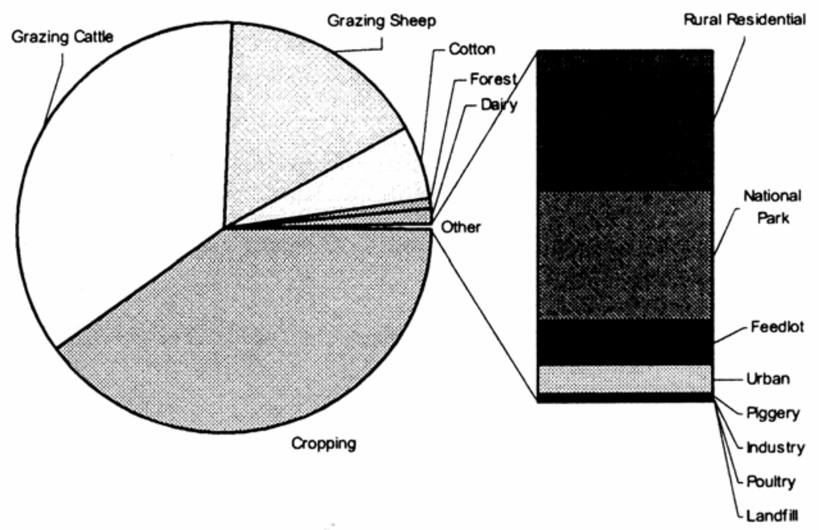


FIGURE 6 – TOTAL PHOSPHORUS CONTRIBUTION BY LAND USE IN THE CONDAMINE BALONNE CATCHMENT (PRENTICE AND WALKER, 2002).

10. SITE VULNERABILITY ASSESSMENT

This section contains details for assessing the state of the farm and its ability to handle effluent and manure reuse. It feeds into the risk assessment process (Section 11) used to evaluate the risk of adverse environmental impacts from reuse. The aim is to provide information to decide vulnerability classes (high, medium or low) for natural resources.

Good design and management practices can sometimes be used to reduce the vulnerability of natural resources. Section 7.5 details a number of practices that can be used to reduce nutrient export from reuse areas. These include:

- Locating vegetative filter strips downslope of the reuse area to reduce the vulnerability of nearby surface waters.
- Locating terminal ponds downslope of reuse areas to reduce the vulnerability of nearby surface waters.
- Installing contour banks on sloping land to reduce soil erosion and the subsequent vulnerability of nearby surface waters.
- Maintaining continuous ground cover land to reduce soil erosion and the subsequent vulnerability of nearby surface waters.
- Using sound reuse practices to minimise effluent runoff and deep drainage of nutrients before plants can use them.

These factors are considered when evaluating the vulnerability of each resource. Since different reuse areas on a property have different risk levels depending on site, design and management factors, the site vulnerability assessment needs to be applied separately to each reuse area. A separate reuse area is any area used for spreading effluent or manure that has a different soil type, land use, by-product type (e.g. composted manure V fresh manure), application method or application rate from other areas. For instance, the effluent reuse area might have a high risk level, while the solids area might pose a low risk.

10.1. Soil

The suitability of the soil for effluent and solids reuse depends on a range of factors. Ideally, reuse area soils should have the following properties:

- Loam to medium clay texture (Heavy clay soils require careful management to avoid irrigation runoff and waterlogging).
- Moderately deep to deep.
- Not subject to erosion.
- Well drained.
- Flat to gently sloping.
- Slightly alkaline to slightly acidic pH.
- Suitable for growing pastures (cut and cart) or forage crops.

Less desirable soils have the following properties:

- Sandy.
- Heavy sodic or dispersive clay texture.
- Shallow depth.
- Erosion prone.
- Poorly drained.
- Slope of over 10% (However, inappropriate slope can be overcome via design and management of the system).
- Subject to structural problems e.g. crusting, slaking, hardsetting and dispersible soils.
- Sodic.
- Saline.
- Elevated nutrient concentration.
- Moderately to strongly alkaline or moderately to strongly acidic pH.

The information given below can be used to determine the vulnerability class of the soil. Some soils will have low vulnerability for some properties and high vulnerability for other properties. The highest vulnerability class across all properties should be assumed to be the overall vulnerability class.

Texture

Low vulnerability: Soil texture is loam to medium clay.

Medium vulnerability: Soil texture is duplex with a light topsoil and a heavy subsoil <u>or</u> is heavy clay.

High vulnerability: Soil texture is sand <u>or</u> unknown.

Depth

Low vulnerability: Depth of soil is > 1 m.

Medium vulnerability: Depth of soil is 0.5 – 1m.

High vulnerability: Depth of soil is < 0.5 m <u>or</u> unknown.

Slope

Low vulnerability: Slope is < 5% <u>or</u> slope is 5-10% but continuous vegetative cover is constantly maintained over the area <u>or</u> slope is 5-10% but a system of well-designed contour banks is in place to slow the movement of water from the site.

Medium vulnerability: Slope is 5 – 10% <u>or</u> slope is >10% but continuous vegetative cover is constantly maintained over the area <u>or</u> slope is >10% but a system of well-designed contour banks is in place to slow the movement of water from the site.

High vulnerability: Slope is > 10% <u>or</u> unknown.

Soil Dispersion

Low vulnerability: Soil does not disperse on wetting and has a low exchangeable sodium percentage (less than 6%).

Medium vulnerability: Soil disperses on wetting and / or has an exchangeable sodium percentage of 6-15%.

High vulnerability: Soil disperses on wetting and / or has an exchangeable sodium percentage exceeding 15% <u>or</u> the dispersive behaviour and exchangeable sodium percentage of the soil are unknown.

Salinity

Low vulnerability: Soil is in the very low to low salinity class (EC_{se} is less than 1.9 dS/m)

Medium vulnerability: Soil is in the medium salinity class (EC_{se} is 1.9-4.5 dS/m)

High vulnerability: Soil is in the high to extreme salinity class (EC_{se} is over 4.5 dS/m) <u>or</u> soil salinity class is unknown.

Nutrient Status

<u>Nitrogen</u>

Low vulnerability: Either soil solution nitrate-N levels at the base of the active root zone are <10 mg/L or are less than measured baseline data.

High vulnerability: Either soil solution nitrate-N levels at the base of the active root zone are >10 mg/L or are greater than measured baseline data.

These can be converted to soil nitrate-nitrogen concentrations for different soil types as per Table 31

<u>Phosphorus</u>

Vulnerability ratings for phosphorus are based on three methods.

Method 1 involves a check as to whether the Colwell Extractable phosphorus levels exceed certain limits. These limits are based on measured Colwell extractable phosphorus for numerous soils (categorised by clay content and pH). The upper limits (high rating) are one standard deviation above the mean of numerous Colwell extractable phosphorus levels (Redding per comms., 2002). However, these limits may not be appropriate for some soil types, such as black vertosols, which may have high levels of Colwell phosphorus in their 'virgin' state.

Method 2 uses guideline limits specifically for acid soils. Some acid soils may require methods involving acid extraction to measure available phosphorus (common in southern NSW and coastal soils). Thus method 2 involves a check as to whether BSES or Bray phosphorus levels exceed certain limits.

Method 3 is an alternative method to 1 and 2 and involves measuring extractable phosphorus levels (with the appropriate method) in the reuse areas and comparing these to extractable phosphorus levels in background plots that have not received effluent or solid by-products.

Method 1 (Most Soils)

Low vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	< 15
< 30%	> 7	< 30
> 30%	< 7	< 40
> 30%	> 7	< 45

Medium vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	15 – 30
< 30%	> 7	30 – 60
> 30%	< 7	40 – 75
> 30%	> 7	45 – 85

High vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	> 30
< 30%	> 7	> 60
> 30%	< 7	> 75
> 30%	> 7	> 85

Method 2 (Acid Soils)

Low vulnerability: Bray phosphorus level < 20 mg/kg
BSES phosphorus level < 15 mg/kg for soils with < 30% clay
BSES phosphorus level < 65 mg/kg for soils with > 30% clay

Medium vulnerability: Bray phosphorus level between 20 and 25 mg/kg
BSES phosphorus level 15 - 30 mg/kg for soils with < 30% clay

	BSES phosphorus level between 65 - 130 mg/kg for soils with > 30% clay
High vulnerability:	Bray phosphorus level > 25 mg/kg BSES phosphorus level > 30 mg/kg for soils with < 30% clay BSES phosphorus level > 130 mg/kg for soils with > 30% clay

Method 3 (Alternate Method to 1 and 2)

Firstly, obtain baseline available phosphorus levels for the soil on an area that has not received effluent or solids. The extraction method will usually be bicarbonate (e.g. Colwell), but in some cases may be an acid extraction. Then measure extractable phosphorus levels in the reuse area.

Low vulnerability	The extractable phosphorus level of the reuse area is less than 150% of baseline data. (Thus if baseline data indicates the level is 30 mg/kg, the trigger level is less than 45 mg/kg).
Medium vulnerability	The extractable phosphorus level of the reuse area is between 150% and 200% of baseline data. (Thus if baseline data indicates the level is 30 mg/kg, the trigger level is between 45 mg/kg and 60 mg/kg).
High vulnerability	The extractable phosphorus level of the reuse area is more than 200% of the baseline data. (Thus if baseline data indicates the level is 30 mg/kg, the trigger level is greater than 60 mg/kg).

If it can be shown from the baseline data that the soil is phosphorus deficient, then the baseline data can be adjusted to 'desirable' phosphorus levels for that particular soil type.

10.2. Surface Water

Overtopping of effluent treatment systems needs to be minimised to protect surface waters. This document only covers the re-use of effluent and solids, however it is acknowledged that the effluent re-use area is linked to the wet weather storage. Thus it is recommended that where appropriate, effluent treatment systems be designed to hold effluent in a 90th percentile wet year for high strength effluent (total nitrogen > 100; total phosphorus > 20) and a 75th percentile wet year for medium strength effluent (total nitrogen 50 - 100; total phosphorus 10 - 20). These criteria vary between states (e.g In Queensland the treatment system should be designed so that it does not overtop more than once every 10 years on average).

Surface water includes water in dams, reservoirs, rivers, creeks and all other waterways where rainfall is likely to collect. Ideally, reuse areas should be well separated from surface water bodies, particularly those used for sensitive purposes e.g. town water supplies. However, distance is not the only criterion determining the potential for contamination from reuse areas. Design and management factors, particularly the amount and type of vegetative cover may significantly reduce any potential contamination of surface waters.

Water Quality Protection

Low vulnerability: Reuse area is at least 200 m from a surface water body and effluent irrigations do not cause runoff or is at least 150 m from a surface water body but includes a vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff or is at least 100 m from a surface water body but includes a well-maintained vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff or there is a terminal pond sized to catch the first 12 mm of rainfall runoff plus irrigation water runoff.

Medium vulnerability: Reuse area is between 100 m and 200 m from a surface water body and effluent irrigations do not cause runoff or is at least 75 m from a surface water body but includes a vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff or is at least 50 m from a surface water body but includes a well-maintained vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff.

High vulnerability: Reuse area has no vegetative buffer and is less than 100 m from a surface water body or reuse area has a vegetative buffer but is within 50 m of a surface water body or and effluent irrigations create runoff that is not captured in a terminal pond.

Flood potential

Low vulnerability: Reuse area is above the 1 in 10 year flood line.

Medium vulnerability: Reuse area is above the 1 in 5 year flood line but below the 1 in 10 year flood line.

High vulnerability: Reuse area is below the 1 in 5 year flood line or flooding frequency of reuse area is unknown.

10.3. Groundwater

Ideally, reuse areas should be located on areas with deep groundwater or on those well protected by a layer of clay or be a confined aquifer. The risk to groundwater from effluent reuse depends upon the protection afforded by soil type (e.g. a deep clay blanket may afford good protection, a sandy loam soil provides relatively poor protection) and the geology and type of aquifer (e.g. a confined aquifer versus an alluvial aquifer).

The consequences of nutrient or salt leaching to groundwater depend on the quality of the groundwater (e.g. potable water V brackish water). However, re-use practices should not impact on groundwater resources since it is this generation's responsibility to protect groundwater quality for the benefit of future generations.

Depth to groundwater

Low vulnerability: Groundwater is at least 20 m below the surface.

Medium vulnerability: Groundwater is 10 - 20 m below the surface.

High vulnerability: Groundwater is less than 10 m below the surface or depth to groundwater is unknown.

Soil type

Low vulnerability: There is at least 0.5 m of clay above the aquifer or the aquifer is confined.

Medium vulnerability: There is at least a metre of loam to clay-loam soil above the aquifer.

High vulnerability: Any other

Water quality

Low vulnerability: The groundwater resources in the area are of a quality having no productive use e.g. EC exceeds 8 dS/m.

Medium vulnerability: Groundwater resources are suitable for stock drinking water or irrigation e.g. EC of up to 8 dS/m & containing less than 100 mg NO_3N/L

High vulnerability: Groundwater resources are suitable for human consumption. (EC of up to 1.6 dS/m and containing less than 10 mg NO_3N/L) or the quality of groundwater resources is unknown.

Table 51, Table 52 and Table 53 are templates for recording the site vulnerability risk weightings for soil, surface water and groundwater. To complete the tables, a vulnerability weighting of 1, 2 or 3 applies to each sub-category of soil, surface water and groundwater. A low vulnerability attracts a vulnerability weighting of “1”, medium vulnerability attracts a vulnerability weighting of “2” and high vulnerability attracts a vulnerability weighting of “3”. These numbers are transferred to Table 54, Table 55 and Table 56 in Section 0.

10.4. Risk Assessment Tables

TABLE 51 – VULNERABILITY WEIGHTINGS - SOIL

Resource	Texture (weighting low = 1, med. = 2, high = 3)	Depth (weighting low = 1, med. = 2, high = 3)	Slope (weighting low = 1, med. = 2, high = 3)	Soil Dispersion (weighting low = 1, med. = 2, high = 3)	Salinity (weighting low = 1, med. = 2, high = 3)	Nitrogen (weighting low = 1, med. = 2, high = 3)	Phosphorus (weighting low = 1, med. = 2, high = 3)
Site Vulnerability Weighting						e.g. 2	

TABLE 52 – VULNERABILITY WEIGHTINGS – SURFACE WATER

Resource	Water Quality Protection Weighting (low = 1, medium = 2, high = 3)	Flood Potential Weighting (low = 1, medium = 2, high = 3)
Site Vulnerability Weighting		

TABLE 53 – VULNERABILITY WEIGHTINGS - GROUNDWATER

Resource	Depth to Groundwater Weighting (low = 1, medium = 2, high = 3)	Soil Type Weighting (low = 1, medium = 2, high = 3)	Water Quality Weighting (low = 1, medium = 2, high = 3)
Site Vulnerability Weighting			

Transfer these values to the Risk Assessment Matrix Tables (Table 54, Table 55 and Table 56).

11. THE RISK ASSESSMENT PROCESS

11.1. Introduction

This risk assessment process considers the site assessment, the whole farm mass balance, the design and management of the reuse area and the sustainability indicators to decide if adverse environmental impacts are likely. The outcome of the risk assessment process is a risk appraisal for each resource and targeted environmental monitoring to measure sustainability.

In determining the level of risk of a reuse practice, the general principles of sustainable effluent irrigation and manure spreading need to be considered, such as those listed in the NSW EPA Draft Use of Effluent in Irrigation Guidelines. These principles are:

Resource Use: Potential resources in effluent, such as water, plant nutrients and organic matter, should be identified, and agronomic systems developed and implemented for their effective use.

Protection of Lands: An effluent irrigation system should be ecologically sustainable. In particular, it should maintain or improve the capacity of the land to grow plants, and should result in no deterioration of land quality through soil structure degradation, salinisation, water logging, chemical contamination or soil erosion.

Protection of Groundwater: Effluent irrigation areas and systems should be located, designed, constructed and operated so that the current or future beneficial uses of groundwater do not diminish as a result of contamination by the effluent or run off from the irrigation scheme or changing water tables.

Protection of Surface Waters: Effluent irrigation systems should be located, designed, constructed and operated so that the surface waters do not become contaminated by any flow from irrigation areas, including effluent, rainfall run off, contaminated sub-surface run off, or contaminated groundwater**.**

Prevention of Public Health Risk: The effluent irrigation scheme should be sited, designed, constructed and operated so as not to compromise public health. In this regard, special consideration should be given to the provision of barriers that prevent human exposure to pathogens and contaminants.

Community Amenity: The effluent irrigation system should be located, designed, constructed and operated to avoid unreasonable interference with any commercial activity or the comfortable enjoyment of life and property off-site, and where possible to add the amenity. In this regard, special consideration should be given to odour, dust, insects and noise.

In addition, an environmental management plan (EMP) or an environmental management system (EMS) will help to assess the environmental risk of an enterprise and any potential environmental impacts will hopefully be addressed. This could be used to provide informed decisions on the level of monitoring needed for a particular enterprise, with a possible reduction in monitoring requirements. An EMP or EMS should provide more information on the level of risk associated with the system, but wouldn't be the only means of determining an appropriate level of monitoring. The level of influence would be determined by the quality of information they contain.

A matrix has been developed to help determine the risk that each effluent or solid by-product area poses to surface water, groundwater and soil. Since different reuse areas on a property have different levels of risk depending on site, design and management factors the matrix needs to be applied to each reuse area. A separate reuse area is any area used for spreading or effluent or manure that has different soil type, land use, by-product type (e.g. composted manure V fresh manure), application method or application rate. For instance, the effluent irrigation area might have a high risk, while the solids area might pose a low risk. Consequently, more stringent monitoring would be needed for the effluent area compared with the solids area.

When interpreting monitoring data there will be considerable variations due to climatic conditions (e.g. wet years, drought) and subsequent effects on crop yields and therefore nutrient uptakes, cropping regime (rotations) and general soil dynamics. Thus, monitoring data should be viewed in terms of trends in the context of the forward management plan (10 – 15 years), which is regularly reviewed (every 3 – 5 years). Single monitoring points that exceed trigger levels do not signify an unsustainable system. Averages or trends (3 – 5 years) need to be used to assess sustainability, with the view of utilising all the nutrients applied in the long term. This includes the utilisation of stored phosphorus after re-use has ceased.

11.2. Risk Assessing the Site

The following matrix combines the site vulnerability assessment with the design and management risk assessment to provide an overall risk assessment of effluent and solids reuse. Transpose information from Section 6.7, Table 30, Table 51, Table 52 and Table 53 into Table 54, Table 55 and Table 56 to complete the matrix.

Multiply each site vulnerability weighting by each Design and Management Risk Weighting to obtain an overall risk assessment for the site (see example in Table 54). The overall level of risk calculated for each site resource (soil, surface water and groundwater) is used to design the appropriate monitoring (targeted monitoring) or improvement.

Risk weighting of 1, 2, 3, 4, 6 & 9 are possible. Ratings of 1 and 2 require minimal monitoring and/or change to design and management. Ratings of 3, 4 & 6 attract moderate levels of monitoring and/or changes to design and management. A rating of 9 requires intensive monitoring and/or changes to design and management. It is important to realise that if a rating of 4 is calculated for groundwater and a rating of 9 is calculated for soil, moderate monitoring and/or change would be warranted for the groundwater and intensive monitoring and/or change would be warranted for the soil.

It is recommended that the risk assessment process be trialed prior to implementation. Ideally, this trialing should include a range of case studies on theoretical and real case piggeries and feedlots to demonstrate how the assessment process would work and the outcomes that it would deliver in terms of the assessed risk and the resultant monitoring requirements. The proposed risk assessment process should be evaluated by applying it to some existing licensed piggeries and feedlots.

Theoretical example risk assessments for a piggery and two feedlots can be found in Appendix G, Appendix H and Appendix I.

TABLE 54 - RISK ASSESSMENT MATRIX - SOIL

Design and Management Criteria Design & Management Risk Weighting	(Low = 1, medium = 2, high = 3)	Texture	Depth	Slope	Soil Dispersion	Salinity	Nitrogen	Phosphorus
		Number from Table 51	Number from Table 51	Number from Table 51	Number from Table 51	Number from Table 51	Number from Table 51 (2)	Number from Table 51
Nutrients in manure and effluent	Number from section 6.7							
Size of land area and Application rate	Number from Table 30 (3)						2	
Application method	Number from Table 30							

TABLE 55 - RISK ASSESSMENT MATRIX – SURFACE WATER

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Site Vulnerability Weighting	
		Water Quality Protection	Flood Potential
		Number from Table 52	Number from Table 52
Nutrients in manure and effluent	Number from Section 6.7		
Size of land area and Application rate	Number from Table 30		
Application method	Number from Table 30		

TABLE 56 - RISK ASSESSMENT MATRIX - GROUNDWATER

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Site Vulnerability Weighting		
		Depth	Soil Type	Water Use
		Number from Table 53	Number from Table 53	Number from Table 53
Nutrients in manure and effluent	Number from section 6.7			
Size of land area and Application rate	Number from Table 30			
Application methods	Number from Table 30			

Based on the Risk Rating from Table 54, Table 55 and Table 56, an evaluation of the likely amount of monitoring and/or change to the design and management that would be required can be determined (See Section12).

12. TARGETED MONITORING

Table 54, Table 55 and Table 56 (Section 11.2) identify the level of overall risk to soils, surface water and groundwater, respectively. Monitoring and/or improved design and management should be undertaken in accordance with the risk level.

Detailed information on the collection, storage, handling and treatment of samples for soil, effluent, solids, surface water and groundwater can be found in Appendix F. The level of sampling suggested is designed to provide valid results, without being cost prohibitive.

When monitoring is used to observe trends, it is worth noting that considerable variations can be obtained via the sampling method and laboratory used for analysis. In addition, time of sampling is important. Soil samples should be collected at the end of the main growing season when the plants have had time to take up the applied nutrients.

12.1. Soils

Where the risk of soil related impacts is low (rating of 1-3) and at least 3 years of annual monitoring shows that the system is sustainable, it is suggested that soils from reuse areas should be monitored at least every three years. Those in a low risk category will not need to monitor effluent quality *unless* they are already undertaking this monitoring (which is the reason for being in this category).

Where there is a medium risk of soil impacts (rating of 4 or 6) and at least 3 years of monitoring data show that the system is sustainable, it is suggested that soils from reuse areas should be sampled and analysed at least every two years. Effluent and solids quality (if reused on-site) should also be analysed annually.

Where there is a high risk of soil impacts (rating of 9), annual soil monitoring is imperative. Effluent and solids quality (if reused on-site) should also be analysed annually.

Table 57 includes recommended soil monitoring parameters. The monitoring results should be compared with the limits for sustainability indicators given in Section 8. Where the triggers for further investigation are reached, further analysis is needed. Table 58 and Table 59 include recommended effluent and solids monitoring parameters.

The quantity of effluent and solids applied to land will need to be measured by everyone, except those relying on a mass balance calculation to demonstrate sustainability.

Crop yields will need to be measured by everyone, except those relying on a mass balance calculation that shows that they are sustainable.

12.2. Surface Water

Surface water quality monitoring is not suggested as a relevant measure of sustainability for piggeries and cattle feedlots, as they are not direct discharge industries (e.g. sewage treatment plants) and generally rely on land application for the reuse of by-products. To be able to achieve any meaningful results from a monitoring perspective, surface water monitoring would require sophisticated equipment and trained operators.

Piggeries and cattle feedlots are required to comply with relevant codes of practice for their design and management, such as appropriate buffers, vegetative filter strips or terminal ponds. If an enterprise attracts a high rating, remedial action in the form of improved design and/or management of the reuse area is warranted.

12.3. Groundwater

Groundwater quality monitoring would be warranted for anyone attracting a high rating (9). Ideally this would include sampling and analysis from bores upslope and downslope of reuse areas. Electrical conductivity and nitrate-nitrogen should be determined. On very sandy soils, total P should also be measured. If a moderate risk weighting is attracted for groundwater, monitoring would not be required, provided nutrient and salt risk weightings for the soil are low.

TABLE 57 – RECOMMENDED SOIL ANALYSIS PARAMETERS

Soil test parameter	Depth (Down profile)	Justification
pH		Influences nutrient availability
EC_{se} (Can measure $EC_{1:5}$ and convert to EC_{se})$^{+}$	0.0 – 0.1 m 0.2 – 0.3 m 0.5-0.6 m OR base of root zone	Measure of soil salinity
Nitrate-N	0-0.1 m 0.2-0.3 m 0.5-0.6 m OR base of root zone	Measure of nitrogen available for plant uptake
Available phosphorus (Colwell or Olsen or Bray or BSES or Lactate or Calcium Chloride or Other)	0-0.1 m 0.5-0.6 m OR base of root zone*	Measure of phosphorus available for plant uptake
P sorption capacity or phosphorus Sorption Index	0 –0.6 m OR 0 – base of root zone**	Essential if applying more than plant uptake
Organic Carbon	0-0.1 m	Influences soil stability and consequently soil erosion
Exchangeable cations and CEC (Calcium, sodium, potassium, magnesium).	0-0.1 m 0.5-0.6 m or base of root zone	Needed to calculate ESP, EKP and Ca: Mg which have important implications for soil structure

$^{+}$ EC_{se} levels in the top soil layers is not intended to be a direct sustainability indicator, but will provide useful agronomic information and provide a guide to soil salt movements.

* Only check available P levels annually at 0.5 – 0.6 m (or base of root zone) if a sandy soil, otherwise every 5 years.

** Measurement of P sorption capacity to 0.6 m or the base or the root zone is desirable before reuse and every 5 years after initial application.

Measuring chloride as 50 – 60 cm (or base of root zone) may also be warranted if further investigations of salinity are required.

TABLE 58 – RECOMMENDED EFFLUENT ANALYSIS PARAMETERS

Test parameter	Justification
Total-N or TKN	Measure of nitrogen applied for mass balance calculations
Ammonium-N	Measure of nitrogen available or potentially lost as ammonia volatilisation
Total P	Measure of phosphorus applied for mass balance calculations
Electrical conductivity and Chloride	Measure of effluent salinity
Sodium Adsorption Ratio (SAR)	Measure of effluent sodicity

TABLE 59 – RECOMMENDED SOLIDS ANALYSIS PARAMETERS

Test parameter	Justification
Dry Matter	To calculate nutrient applied
Total-N or TKN	Measure of nitrogen applied for mass balance calculations
Ammonium-N	Measure of nitrogen available or potentially lost as ammonia volatilisation
Nitrate-N	Measure of nitrogen immediately available for plant uptake
Total P	Measure of phosphorus applied for mass balance calculations
Organic Carbon	Influences soil stability
Electrical conductivity and Chloride	Measure of effluent salinity

13. REVIEW OF FORWARD MANAGEMENT PLAN

Where interpretation of the monitoring results and/or the risk assessment identifies a need to improve performance, the **Forward Management Plan** would be reviewed. This is the stage where the design and management of the reuse system is evaluated to find ways to reduce the potential risk to the environment. Once changes are implemented, the risk assessment process must be repeated to decide the new level of risk and the appropriate monitoring regime to complement the revised level of risk.

14. CONCLUSIONS AND RECOMMENDATIONS

This section details conclusions from the study and highlights gaps in information related to effluent and solids reuse for piggeries and cattle feedlots. There is currently a significant amount of work being undertaken both in Australia and overseas. It is anticipated that the findings of these studies will improve the general understanding of reuse. Recommendations are also made for possible future research to better understand the processes.

The study has identified sustainability indicators for a number of parameters: nitrogen, phosphorus, salinity and sodicity. For these sustainability indicators, trigger values have been identified to assist industry in reviewing their effluent and manure reuse forward plans. The monitoring and review of performance using these sustainability indicators will assist industry with operating environmentally sustainable operations.

Following are recommendations for the sustainability indicators:

- For nitrate-nitrogen, a limit of 10 mg/L below the active root zone is suggested only as a trigger for further investigation. For nitrogen, sustainability of reuse practices depends also on the risk of nitrate moving off-site in stormwater runoff and by leaching to groundwater, the quality (value) of the groundwater and the amount of deep drainage of the soil of the reuse area. These need to be evaluated as part of the risk assessment of the reuse area.
- For phosphorus, it is recommended that storage of phosphorus be allowed based on the calculated storage capacity from the phosphorus sorption isotherm, at a soil solution concentration of 0.5 mg/L. However, this soil solution concentration needs review as the recent work of Redding and others emerges. Other soil solution concentrations may be appropriate for different soil types and regions depending on available data. Another test that offers potential is the simple test for estimating phosphorus buffer capacity (PBC) that was developed by Burkitt *et al.* (2002). Their methods provide a simple and accurate method for estimating PBC. However, this work requires further evaluation to ascertain whether their data can be used to provide simple indices for determining phosphorus sustainability of a range of soil types, not only in NSW, but for the cropping soils of Australia in general.

A risk assessment process has also been developed. This risk assessment process considers the site assessment, the whole farm nutrient mass balance, the design and management of the reuse area and the sustainability indicators to decide if adverse environmental impacts are likely. The outcome of the risk assessment process is a risk appraisal for each resource and targeted environmental monitoring to measure sustainability.

No recommendations are made concerning the application (or not) of Load Based Licensing to piggeries and cattle feedlots and the application of the currently existing Load Calculation Protocol to piggeries and cattle feedlots. This process needs to be negotiated between the industries involved and the NSW EPA.

Currently licensed piggeries and cattle feedlots in NSW have collected significant monitoring data. This collected information could be used to trial the developed risk assessment process. As part of this current study, three theoretical risk assessments have been completed to further explain how the process would work. A further trial of the risk assessment process could include a range of case studies on real piggeries and feedlots to demonstrate how the assessment process would work and the outcomes that it would deliver in terms of the assessed risk and the resultant monitoring requirements. This would allow the process to be properly evaluated for both the piggery and feedlot industries. Theoretical example risk assessments for a piggery and two feedlots can be found in Appendix G, Appendix H and Appendix I.

The Load Calculation Protocol proposes a 15 year forward management plan with a review of the plan every 3 years to ensure that future planned application rates will continue to achieve sustainable assimilation. FSA Environmental agrees that there is a need for a plan for managing nutrients for reuse. Review via monitoring results at least every three years is necessary to judge performance. Plans for proposed reuse should consider monitoring results. Whether a 15 year forward management plan is strictly needed is debatable. The main priority should be a forward plan that is regularly reviewed and updated in light of monitoring results.

The cattle feedlot industry agrees with the 15 year forward management plan. We recommend that if the pig industry wishes, they adopt a 5 – 10 year forward management plan that is regularly reviewed.

The general recommendations for sustainable reuse presented in the report apply to most industries that reuse their by-products in a land application system. However, inherent differences will apply for industries that generate larger volumes of water compared to piggeries and cattle feedlots. These, and any other differences would need to be evaluated when considering the application of these sustainability indicators to other industries.

It is recommended that EPA review their monitoring requirements for piggeries and cattle feedlots. The level of monitoring required should be based on the level of environmental risk as determined by the risk assessment process. The level of environmental risk specific should also determine the parameters measured.

The authors believe that a defined path for upgrading Codes of Practice and Guidelines is lacking, with many of these documents being outdated and/or very conservative because of a lack of knowledge ‘Precautionary Principle’. This is however, changing with many of the more recently produced codes for intensive animal industries planning 5-year reviews and upgrades of the publications. It would be beneficial that as part of current and future research (APL and MLA), relevant, peer reviewed findings be included in regular upgrades of codes and guidelines. This is most likely to be successful if national codes and guidelines exist for the industries. This process can proceed, as the feedlot industry currently has a National Code and the pig industry is developing a National Guideline.

The sustainability indicators identified from this project are considered to be the best available at the time of project completion. However, due to the significant work being undertaken currently in this area, particularly by the piggery and feedlot industries, it is recommended that they be regularly reviewed to ensure they remain relevant.

The authors advise that it is important that the NSW EPA and operators of piggeries and cattle feedlots recognise that it is extremely difficult to develop tools for determining and demonstrating sustainability and indicators of sustainability that adequately cover all situations. It is probable that situations will arise where the tools for determining sustainability overstate the likely risk to the environment. Similarly, while the best-bet indicators of sustainability have been identified in this project, these may occasionally provide an inaccurate assessment of environmental impact. Consequently, where a significant level of environmental risk or impact is identified, it is critical to confirm that the result is accurate through further examination.

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Appendix A. NITROGEN: ANALYSIS METHODS & SAMPLING

Analysis Methods (Including Accuracy)

Strong and Mason (1999) state that "Since nitrogen is either taken up from the soil as either nitrate or ammonium, the supply of these soil mineral nitrogen forms has been used to derive a measure of the plant's available nitrogen supply at a strategic time during cropping, which for an annual crop may be prior to sowing. Because of the various sources of plant-available nitrogen in soil and the dynamic nature of the supply, snapshots in time (soil tests) may not successfully predict the supply of plant-available nitrogen".

Rayment and Higginson (1992) state that "From a land use/soil fertility viewpoint, the total nitrogen and 'mineral nitrogen' components are of particular interest. Nitrate, and to a lesser extent ammonium, are important sources of nitrogen for plant growth, while total nitrogen provides a measure of the quantity of nitrogen that can be 'mineralised' under appropriate conditions".

Total Nitrogen

"Measurement of total nitrogen, based on wet oxidation (Kjeldahl 1883) has found wide acceptance" (Bremner and Mulvaney, 1982, as cited in Rayment and Higginson, 1992). The Kjeldahl digestion (see below) is modified to ensure the recovery of NO_3-N and NO_2-N. "Nitro compounds are initially formed when soils containing NO_3-N react with salicylic acid/H_2SO_4. These nitro compounds are subsequently reduced to corresponding amino compounds by heating and mixing with sodium thiosulfate ($Na_2S_2O_3.5H_20$) prior to conventional Kjeldahl digestion/distillation" (Rayment and Higginson, 1992).

TKN

Total Kjeldahl nitrogen is the sum of the ammonia nitrogen and organic nitrogen in the sample. It does not account for nitrogen in the form of azide, azine, azo, hydrazone, nitrate, nitrite, nitrile, nitro, nitroso, oxime, and semi-carbazone. There are 2 Kjeldahl methods- macro and semi-micro. They operate on the same principle but differ in volume and digestion apparatus. The major factor that influences the selection of macro- or semi-micro-Kjeldahl method is the concentration of organic nitrogen. The macro-Kjeldahl method is applicable for samples containing either low or high concentrations of organic nitrogen but requires a large sample volume for low concentrations. In the semi-micro- method, the sample volume should be chosen to contain Kjeldahl nitrogen (organic nitrogen and ammonia nitrogen) in the range of 0.2 to 2 mg/L. These methods are described in Rayment and Higginson (1992).

Mineral N

Rayment and Higginson (1992) suggest "The measurement of NH_{4-}^{+}-N and NO_3^{-}-N in soils must be undertaken with caution, since rapid transformations can alter their apparent concentrations. Water or salt solutions are suitable extractants for NO_3^{-}-N (and NO_2^{-}-N) in the majority of soils. As an exchangeable base, NH_{4-}^{+}-N must be displaced from the surface of negatively charged soil colloids with another cation, commonly by K^+ or Na^+. Once extracted, NH_{4-}^{+}-N, NO_3^{-}-N and NO_2^{-}-N can be determined by steam distillation/titration or by automated colorimetric procedures".

Rayment and Higginson (1992) describe the extraction procedure of mineral nitrogen from soil as: "……. 2M KCl at a 1:10 soil/solution ratio for 1 h. Magnesium oxide is used for the distillation of NH_3/NH_4^+-N. Finely ground Devarda's alloy is employed for the reduction of NO_3^--N and NO_2^--N to NH_4^+-N while sulfamic acid is added when necessary to destroy NO_2^--N. The NH_4^+-N present or formed is steam distilled into a H_3BO_3 solution and the amount present calculated following titration with dilute mineral acid to pH 5.0. Depending on how the method is utilised, it is possible to measure the following mineral-N fractions: NH_4^+-N, NO_3^--N + NO_2^--N, NH_4^+-N + NO_3^--N + NO_2^--N, NH_4^+-N + NO_3^--N, NO_3^--N. NO_2^--N can be calculated by difference". The interested reader is directed to Rayment and Higginson (1992) for a full description of the individual procedures.

Which is the Best Soil Nitrogen Test

From a number of different studies, Strong and Mason (1999) concluded that for cereal soils of Queensland and northern New South Wales, extraction of nitrate nitrogen with KCl has been found to be useful in determining its contribution to plant-available nitrogen. "Other measurements of the soil's capacity to increase the inorganic nitrogen supply have been devised by using either a chemical extractant or a biological process to estimate nitrogen release rate" (Strong and Mason, 1999).

Strong and Mason (1999) state "Several soil chemical extractants predict either the quantity of mineral nitrogen or the fraction of soil nitrogen which can be released for plant uptake. Methods which use strong chemical extractants (e.g. boiling 6 M HCl or 4.5 M NaOH) usually extract more soil nitrogen than is released biologically during incubation and do not correlate well with plant uptake (Stanford, 1982). Several mild chemical extractants have been found to be useful indices of the nitrogen release rate, including:

- Ammonia released by steam distilling the extract of soil and 0.01 M or 0.02 M $KMnO_4$ in 0.5 M H_2SO_4 (Stanford, 1982).
- Ammonia released on autoclaving soil with dilute $CaCl_2$ (Keeney, 1982).
- Soil digestion for 4 hours with 2 M KCl (Gianello and Bremner, 1986a).
- Steam distillation of soil with phosphate-borate for 8 minutes (Gianello and Bremner, 1986b)."

"While tests involving biological release better simulate the naturally occurring N-release process they are still to be regarded as indices or relative measures of field mineralisation" (Strong and Mason, 1999). Nitrogen mineralisation potential is a test that simulates the quantity of soil nitrogen, which may be released during a growing season. It involves a procedure of accumulating nitrogen released during successive aerobic incubations over a 30 week period" (Stanford and Smith, 1972; Xu *et al.,* 1996, as cited in Strong and Mason, 1999)".

Strong and Mason (1999) conclude that "Soil tests commonly used for N, soil nitrate or tests of the soils capacity to supply mineral N, are unlikely to provide definitive information about the quantity of plant nitrogen that will be available in any cropping system or precise supplementary nitrogen requirements. Monitoring soil nitrate nitrogen during the production/cropping cycle however will continue to assist in nitrogen management to prevent over-supplies of plant-available nitrogen contaminating groundwater".

"Unless special precautions are taken, Kjeldahl digestion does not ensure quantitative recovery of all forms of soil N, especially those with N-O bonds" (Rayment and Higginson, 1992).

Which is the Best Effluent and Solid By-product Nitrogen Test

Effluent from both piggery and cattle feedlot effluent ponds should be tested for both total nitrogen (or total Kjeldahl nitrogen) and mineral nitrogen. The total nitrogen test will provide an indication of the total amount of nitrogen potentially available for plant uptake. A measure of the mineral nitrogen will generally be a measure of the ammonium-nitrogen, as both piggery and cattle feedlot effluent ponds will contain little, if any nitrate-nitrogen and nitrite-nitrogen, because they will generally be anoxic. A measure of this ammonium-nitrogen will provide an indication of how much is potentially available for immediate plant reuse or lost as ammonia volatilisation upon application. Typical feedlot and cattle feedlot effluent irrigation pond concentrations of nitrogen and its forms are shown in Table 10 and Table 21.

As with effluent, solid by-products (screenings, sludge, stockpiled manure, compost and used bedding) should be analysed for total-N or TKN, ammonium-N, nitrate-N.

Appendix B. PHOSPHORUS: ANALYSIS METHODS & SAMPLING

Analysis Methods

This section covers methods for measuring available phosphorus and applicability to different soil types - (standards, costs, accuracy, sample storage & handling).

The analyses of effluent and solid by-products should include both total phosphorus and orthophosphate concentrations.

Total phosphorus

Rayment and Higginson (1992) proclaim that most unfertilised Australian soils contain relatively small amounts of phosphorus, with much of this being immobilised in forms not readily available to plants, such as organically bound and insoluble mineral phosphorus. Thus a measure of the total phosphorus is rarely related to the quantity of phosphorus available to plants.

Table 10, Table 11, Table 21 and Table 22 show typical measured values of total phosphorus and ortho-phosphorus in piggery effluent, piggery sludge, feedlot effluent and feedlot manure respectively.

Extractable phosphorus

"Several factors affect the supply of soil phosphorus to plant roots. These are the amount of phosphorus contained in soil minerals and organic matter, its degree of solubility, the phosphorus concentration of the soil solution, and the rate of diffusion of the plant root surface" (Moody, 1985; as cited in Rayment and Higginson, 1992).

"Because the kinds and amounts of phosphorus in soils are influenced by parent material, weathering processes, vegetation, etc., no single extracting reagent for 'available' soil phosphorus has been formulated" (Thomas and Peaslee, 1973; as cited in Rayment and Higginson, 1992).

"In Australia, several empirical extractants for soil phosphorus are employed. They are the bicarbonate extractions of Colwell (1963) and Olsen *et al.* (1954), lactate-extractable P, fluoride-extractable phosphorus (Bray 1), dilute $CaCl_2$-extractable phosphorus and acid-extractable P" (Rayment and Higginson, 1992).

Moody and Bolland (1999) warn, "In some calibration studies, although the phosphorus test may be described by one of the above methods, the original extracting conditions have been altered. These details should be checked before comparing critical values for the 'same' soil test from different sources"

There is no one single relationship to interpret between phosphorus soil tests. Moody and Bolland (1999) noted, "A few studies have developed relationships between some of the phosphorus soil tests so that, for a given group of soils, one test can be used to predict another. They caution, "These relationships cannot be extrapolated outside the soil types for which they were derived. This is because of the effects of soil chemical properties on such relationships".

Rayment and Higginson (1992) describe analytical methods for determining extractable phosphorus, including the following more common tests:

- Bicarbonate extractable phosphorus – manual colour
- Bicarbonate extractable phosphorus – automated colour
- Olsen-extractable phosphorus – manual colour
- Olsen-extractable phosphorus – automated colour
- Lactate extractable phosphorus – manual colour
- Fluoride-extractable phosphorus (Bray-1P) – manual colour
- Fluoride-extractable phosphorus (Bray-1P) – automated colour
- Acid-extractable phosphorus – *automated colour*

Refer also to Moody and Bolland (1999) for variations of the following tests:

- Bicarbonate extractable phosphorus – manual colour
- Lactate extractable phosphorus – manual colour
- Fluoride-extractable phosphorus (Bray-1P) – manual colour

Phosphate sorption curve

"The relationship between quantity and intensity for a soil is determined by measuring its phosphorus sorption curve (Q/I plot). A phosphorus sorption curve is constructed by adding several rates of soluble phosphorus to a soil suspension and measuring the phosphorus remaining in solution after a suitable equilibration period. The amount of phosphorus added which is adsorbed (Q) is plotted against the concentration of phosphorus remaining in solution (I)" (Moody and Bolland, 1999).

"A standardised phosphorus sorption measurement, which comprises adsorption as well as precipitation reactions is an important soil phosphorus characteristic. Moreover, phosphorus sorption is related to soil phosphorus buffer capacity. With increasing buffer capacity, the proportion of labile phosphorus that is adsorbed by plants, tends to decrease" (Rayment and Higginson, 1992).

"Soils may be grouped together on the basis of their ability to remove and retain phosphorus from a particular solution. Indeed, knowledge of the phosphorus sorption capacity of soil can help in its classification, and in assessing comparative phosphorus fertiliser requirements for plant growth" (Kou *et al.* 1988; as cited in Rayment and Higginson, 1992).

"Sorption curves can be established by the addition of soil of graded amounts of phosphorus 0.01M $CaCl_2$ or 0.02M potassium chloride (KCl). The $CaCl_2$ solution is the more commonly used in Australia (Fox, 1978 and Probert, 1983). The supernatant phosphorus concentration (C) is measured following an equilibration period of 17 h. Calculation of the amount of phosphorus sorbed can then be made" (Rayment and Higginson, 1992).

"The phosphorus sorption curve for a given soil is constructed by plotting phosphorus sorbed against $\log_{10}C$ for each addition of phosphorus. The plot is linear for most soils up to a supernatant phosphorus concentration of at least 0.1 mg P/L" (Rayment and Higginson, 1992).

TABLE 60 – EXTRACTING CONDITIONS FOR VARIOUS SOIL PHOSPHORUS TESTS (MOODY & BOLLAND, 1999)

Method	Extractant	Soil/extract. ratio	Extraction period	Reference
Ammonium lactate-acetic acid	0.1 M ammonium lactate + 0.4 M acetic acid	1:20	30 min	Egner *et al.,* 1960
Bray 1	0.03 M NH_4F in 0.025 M HCl	1:7	60 s	Bray & Kurtz, 1945
Bray 2	0.03 M NH_4F in 0.1 M HCl	1:7	40 s	Bray & Kurtz, 1945
BSES	0.005 M H_2SO_4	1:200	16 h	Kerr & von Stieglitz, 1938
Calcium chloride	0.005 M $CaCl_2$	1:5	18 h	Moody *et al.* 1983
Calcium acetate lactate	O.1 M calcium lactate + 0.01 M calcium acetate + 0.3 M acetic acid	1:20	2 h	Schuller, 1969
Colwell	0.5 M $NaHCO_3$, pH 8.5	1:100	16 h	Colwell 1963
Equilibrium phosphorus concentration	0.01 M $CaCl_2$	1:10	18 h	Moody *et al.* 1983
Fluoride	0.5 M NH_4F	1:50	30 min	Holford *et al.* 1985
Lactate	0.02 M calcium lactate in 0.01 M HCl	1:50	90 min	Holford *et al.* 1985
Mehlich 1	0.05 M HCl + 0.05 M H_2SO_4	1:4	5 min	Mehlich, 1953
Olsen	0.5 M $NaHCO_3$, pH 8.5	1:20	30 min	Olsen *et al.* 1954
P_i	Iron oxide impregnated paper in 0.01 M $CaCl_2$	1:40	16 h	Menon *et al.* 1990; Menon *et al.* 1991
Truog	0.001 M H_2SO_4 + 0.3% $(NH_4)_2SO_4$	1:200	30 min	Truog, 1930

Single point phosphorus sorption indices

"Several 'single point' methods for estimating the phosphorus sorption of soils have been developed, the most widely used being those of Bache and Williams (1971) and Saunders (1965 and 1968)" (Rayment and Higginson, 1992).

Moody and Bolland (1999) suggest, "It is more practical, from a routine analytical viewpoint, to derive an index of phosphorus sorbing ability by using only one phosphorus addition". They have defined 6 sorption categories for these phosphorus sorption indices (Table 61).

TABLE 61 – GENERALISED CATEGORIES OF PHOSPHORUS SORBING ABILITY CORRESPONDING TO VALUES OF SOME COMMONLY USED PHOSPHORUS SORPTION INDICES (MOODY AND BOLLAND, 1999)

Sorption category	$PBC_{0.3}$[A] (mL/g)	PSI_{150}[B]	PRI (mL/g)	Reactive Fe[C] (mg/kg)	NZI[D] (%)	PBC[E]
Very very low	< 10	< 20	< 2	< 100	< 5	< 10
Very low	10 – 50	20 – 30	2 – 20	100 – 300	5 – 10	10 – 20
Low	50 – 100	30 – 40	20 – 50	300 – 500	10 – 20	20 – 30
Moderate	100 – 200	40 – 60	50 – 100	500 – 1000	20 – 40	30 – 60
High	200 – 300	60 – 80	100 – 150	100 – 1500	40 – 60	60 – 90
Very high	> 300	> 80	> 150	> 1500	> 60	> 90

[A] P sorbed x 10 (mg/kg) between soil solution P concentrations of 0.25 and 0.35 mg P/L (Ozanne & Shaw, 1967).
[B] P sorbed (mg/kg)/$\log_{10}$ solution P concentration (µg P/L) for a P addition of 150 mg P/kg (Method 9I1 in Rayment & Higginson, 1992).
[C] P sorbed (mg/kg)/ solution P concentration (mg/L) for a P addition of 200 mg P/kg (P retention index of Allen & Jeffrey, 1990).
[D] Ammonium oxalate extractable Fe (Bolland *et al*. 1996).
[E] New Zealand Retention Index (Blakemore et al. 1987); categories modified from those suggested by Cox (1978).
[f] Slope of P sorbed (mg/kg) vs $\log_{10}$ solution P concentration (µg P/L) (Method 9J1 in Rayment and Higginson, 1992).

Reporting Available P

When a test reports PO_4-P the test is likely to include all of the soluble inorganic phosphorus (orthophosphate), and maybe even some of the organic phosphorus that will become readily available. The test will be colorimetric and will not distinguish between what is PO_4-P and other orthophosphates (Redding pers. comms. 2002).

Equilibrium phosphorus concentration

Magdoff *et al.* (1999) describe equilibrium phosphorus concentration (EPC) as "The solution phosphorus concentration when no adsorption or desorption from the soil occurs. This is the soil solution concentration, or intensity, that will initially be present as phosphorus uptake by roots begins". They found that EPC is strongly related to phosphorus extracted by $CaCl_2$.

When calculating phosphorus concentration at 0.5 mg/L, the following equation can be used:
Concentration sorbed at solution concentration of 0.5 mg/l = PBC*($\log_{10}$(500/EPC))

PBC – Phosphorus Buffer Capacity
EPC – Equilibrium Phosphorus Concentration in ug/L.

What is the Best Phosphorus Soil Test?

Generally when applying organic by-products as fertiliser from piggeries and feedlots, if the amount applied is calculated to match the nitrogen requirements of the plants, phosphorus will be applied in excess of plant requirements. This is generally not a problem as soils have a capacity to store the phosphorus and it will not be released to the environment via leaching and dissolution to surface waters. The capacity of a soil to store phosphorus can be estimated via the phosphorus sorption isotherm. So in terms of defining a sustainability indicator for phosphorus, the amount of phosphorus that is in a form in the soil that can be lost to the environment is the most critical. This may include soluble phosphorus that can be lost via leaching. Also of interest are high phosphorus concentrations in the soil surface that may be lost via erosion. These losses can be minimised through appropriate soil conservation practices (minimum tillage, maintain cover, contours and waterways) and other practices that reduce losses (vegetative filter strips).

Many researchers have investigated the efficacy of different soil tests for different regions and/or soil types. This information is supplied below.

Holford, 1997

Holford (1997) suggests that because of the wide variability in soils and climatic conditions in Australia, there is no one single phosphorus soil test that can be universally applied. The three sets of conditions that apply in Australia are described as:

1. *"Extremely phosphorus deficient soils (e.g. Western Australia agricultural soils)*
2. *Very calcareous nature and/or high pH of many semi-arid soils (e.g. South Australia)*
3. *Moderate or very acid soil conditions (e.g. high rainfall areas of eastern Australia)"*

Due to "*the extreme deficiency of phosphorus in Western Australian soils means that measurements of the very low levels of available phosphorus is relatively unimportant, whereas the reaction of the soil to applied fertiliser may be more important. Hence the phosphorus sorption test may be more useful than a traditional phosphorus extractant test (Ozanne and Shaw 1967)."*

"The highly calcareous nature and/or high pH of many soils will have an important effect on the extracting power of acidic anionic extractants such as lactate and fluoride, which have proved to be the most effective on slightly acidic to alkaline soils of New South Wales. These tests have apparently proved ineffective on calcareous soils of South Australia and Queensland because of the neutralising effect of the soil carbonate on the tests acidity. On the slightly calcareous soils (pH >7.2) soils of New South Wales, the very acidic (pH 1) Bray-2 test has proved much more effective than the moderately acidic (pH 3) Bray-1 test (Holford and Doyle, 1992). The lactate test with a pH of only 3.7, has also proved much more effective than the Bray-1 test because of its wider soil/solution (50:1), which resists the neutralising effect of the carbonate. On soils containing significant amounts of carbonate, it is likely that one of the sodium bicarbonate tests (Olsen or Colwell) would be most effective, as this test was developed specifically for such soils (Olsen et al. 1954)."

Holford (1997) suggests, *"The most effective soil test usually consists of an anionic extractant. Acidic lactate or fluoride have been found most effective in New South Wales, on a range of soils, except calcareous soils which neutralise the acid component (usually hydrochloric and acetic acid) of the extractant. Sodium bicarbonate (pH 8.5) has been found effective on calcareous soils and is widely used throughout the world. It has proved unreliable in NSW soils, and may need more thorough evaluation on non-calcareous soils in other parts of Australia".*

"Although widely used throughout Australia, there is little published evidence of the efficacy or superiority of bicarbonate tests, and they have proved ineffective on wheat-growing soils of NSW which generally contain negligible amounts of carbonate (Holford et al.1985; Holford and Cullis 1985; Holford et. al.1988; Holford and Doyle 1992). By widening the solution/soil ratio and increasing the shaking time of the Olsen bicarbonate test, Colwell (1963) extracted larger amounts of soil P, which was more closely correlated with exchangeable phosphorus and therefore less correlated with phosphorus response. This also means that the critical value for the Colwell test (i.e. the value below which a significant response to fertiliser will occur) will tend to increase as the soil phosphorus sorptivity increases (Holford, 1980b), hence making the test more difficult to interpret."

"No soil tests have been found satisfactory on the very acid, wheat growing soils of southern NSW (Holford and Cullis, 1985b). In very acid soils, adsorbed (exchangeable) phosphorus is held very strongly by soil colloidal surfaces, and it is likely that sparingly soluble mineral phosphorus is more important in replenishing the soil solution than adsorbed phosphorus (Holford, 1983). The Bray-1 test, which extracts Al phosphate as well as adsorbed P, has proved more effective than other tests on acidic pasture soils of the Northern Tablelands of NSW (Holford and Crocker, 1988)."

"On wheat growing soils of central and northern NSW, the lactate and Bray-2 test have proved consistently superior to others. These test give phosphorus values that are well correlated with solution phosphorus concentrations but weakly correlated with exchangeable P, and hence are sensitive to buffering capacity (Holford and Doyle, 1992). Lactate (either Ca or ammonium) is an organic anion, and it is possible that lactate extraction may better simulate the natural mechanism of phosphorus removal (displacement) from soil surrounding the rhizosphere environment of the plant root than other extractants."

"Soils are usually tested for phosphorus for the purpose of estimating their fertiliser requirement for optimum crop or pasture growth. A satisfactory relationship between soil test values and plant phosphorus uptake or fertiliser response does not necessarily indicate a satisfactory predictive relationship between soil test values and fertiliser requirement (Holford et al.1985). Ideally and most simply, there should be a direct predictive relationship between test values and fertiliser requirement, and the latter should be measured in absolute terms rather than relative terms (e.g. fertiliser required for 90% of maximum yield). In practice, absolute fertiliser requirements may be found to be better correlated with soil test values (Holford and Doyle 1993), but there are few published studies giving this type of information."

"Much of the confusion that presently exists concerning the relative efficacy of different soil tests may be removed if the large amount of data from other States, both published and unpublished, were re-evaluated using the same type of approach. The reluctance of soil researchers to re-assess old, but often invaluable, data, and of some soil testing laboratories to discard long-standing but discredited procedures, such as the Colwell test in NSW, are two major hindrances to the advancement and acceptance of soil phosphorus testing practices in Australia today."

Moody and Bolland (1999)

Moody and Bolland (1999) describe the quantity-intensity relationship of phosphorus nutrient availability as *"The quantity-intensity concept of nutrient availability (Schofield, 1955) is useful for understanding the equilibrium between phosphorus in solution and phosphorus in adsorbed or solid phase pools. Quantity (Q) is defined as the amount of phosphorus that is potentially available for plant uptake during a crop cycle. It equates to a portion of the absorbed phosphorus pool and the phosphorus dissolving from fertiliser reaction products during the time-frame of the crop cycle."*

"Intensity (I) is the activity (more simply, the concentration) of inorganic phosphorus in the soil solution. Buffer capacity is the interrelationship between quantity and intensity, is defined as the change in quantity required for a given change in intensity. If we restrict consideration of the Q-I concept to the reactions of inorganic P, then a soil with a high buffer capacity will require more added phosphorus to attain a non-limiting soil solution phosphorus concentration than a soil of lower buffer capacity. Conversely, once the phosphorus concentration of the soil solution of a higher buffer capacity soil has been raised to a non-limiting level, the soil has the ability to maintain that solution phosphorus concentration against depletion by plant uptake for a longer period of time than a soil of lower buffer capacity. Thus, buffer capacity can be of particular importance to phosphorus uptake (e.g. Holford, 1988) and fertiliser requirements (e.g. Dear et al.1992)."

They further analysed the efficacy of various phosphorus soil tests "By considering the extraction parameters (Table 60) for a soil test, it is possible to infer whether it is primarily estimating quantity, estimating intensity, or is a composite index of both (i.e the buffer capacity of the soil is being taken into account (Williams, 1962)). Soils with wide soil to extraction ratios, long shaking periods, concentrated extractant and buffered pH primarily estimate quantity. An example of this is the Colwell soil test. Conversely, soil tests with narrow soil to extractant ratios, short shaking periods, an unbuffered low strength salts estimate intensity. The $CaCl_2$ soil test is an example of this. Many other phosphorus soil tests (e.g. Bray 1 and 2, Olsen, lactate) are intermediate in that they give composite estimates of phosphorus availability that are affected to a greater or lesser extent by the phosphorus buffer capacity of the soil. Perhaps the most versatile method of assessing phosphorus status is the anion exchange membrane (e.g. Simpson et al.1993). By altering the soil-to-membrane-to-solution ratio, the extracting period and the saturating anion, it is possible to derive estimates of either quantity or intensity."

Magdoff et al. 1999

Magdoff *et al.* (1999) reported on four different collections of surface soil samples over a 10 year period. Although this work was conducted in the U.K. on a range of common soils there, including Entisols, Inceptisols, Spodosols and Alfisols it does provide some valuable information on determining the most applicable soil test for phosphorus in terms of plant availability and environmental assessment.

They found that the phosphorus availability to plants, CaCl2-extractable P, and the EPC were all more closely related to ammonium-acetate extractable phosphorus than phosphorus extracted by solutions containing F, such as Mehlich 3, Bray and Kurtz 1, and phosphorus extracted with acetate + F. The fraction of reactive Al that has reacted with phosphorus (as estimated by the ammonium-acetate extractable phosphorus or the ration of phosphorus extracted with acetate + F/ammonium-acetate reactive aluminium) appears to be a better indicator of phosphorus availability and potential phosphorus desorption to runoff water than is phosphorus extracted with F.

The availability of phosphorus to plants was shown to be closely related to the concentration in the soil solution (intensity) and the soil's ability to replenish or buffer the phosphorus concentration as phosphorus is removed by plants (capacity).

They also concluded that it apparently takes extremely high soil test phosphorus levels, perhaps > 120 mg/kg of ammonium-acetate extractable phosphorus to result in runoff phosphorus concentrations >1 mg of P/L. They used the 1 mg/L of dissolved phosphorus limit for point source discharge for agricultural runoff from the US EPA (1986). To put this into perspective, of the approximately 2600 agricultural soils submitted to their lab in 1996, the highest individual sample level was 111 mg/kg of ammonium-acetate extractable P, and the highest 10% averaged 32.8 mg/kg of ammonium-acetate extractable phosphorus. The remaining 90% of samples were below 17 mg/kg of ammonium-acetate extractable phosphorus. They suggested that runoff water with dissolved phosphorus > 1 mg/L would be rare under normal conditions, unless runoff occurred soon after surface phosphorus application.

Redding (2002)

Test should be based on what you are trying to find. e.g If you are only interested in the soluble P, you should use the Calcium Chloride Extraction. This however may be below the detection limit where there has not been a lot of phosphorus applied.

There have been large reported differences in soil Colwell phosphorus test results from different laboratories given split samples by both FSA Environmental and DPI, Queensland. According to Redding (pers comms), some variation may be due to the preparation of the sample for analysis at the laboratory. The fineness of grind of the soil sample it likely to affect the amount of phosphorus extracted using the Colwell test. A grinder that produces a large percentage of particles that are significantly smaller than 2 mm will produce more extractable phosphorus.

Burkitt et al. (2002)

Burkitt *et al.* (2002) has developed a new test that has the potential to provide the necessary information for agronomists and advisers to improve the accuracy of phosphorus fertiliser recommendations and critical Colwell phosphorus extractable adjustment. In their study, the phosphorus sorption indices for an addition of 1000 mg p/kg (PBI_{+ColP} and PBI_{+OlsP}) were found to be superior indicators of phosphorus Buffer Capacity -PBC (Ozanne and Shaw, 1968), in comparison to soil classification, soil texture and other selected soil properties. The ability of PBI_{+ColP} to accurately predict PBC was proven on 34 independent soils, their data indicates that both PBI_{+ColP} and PBI_{+OlsP} are suitable for application across a wide range of Australian soils, regardless of soil type. They claim that this work should lead to an increase in the overall efficiency of phosphorus fertiliser use and thus reducing the risk of phosphorus losses from Australian agricultural systems.

Appendix C. SALINITY: ANALYSIS METHODS & SAMPLING

Measuring Salinity

Electrical conductivity (EC) and total dissolved solids (TDS) are the most common measures of salinity. Total dissolved ions (TDI) is another measure.

EC

EC is a measure of the quantity of electricity conducted by a liquid. It is the reciprocal of electrical resistance and increases with salt concentration. This is because salts dissolving in water disassociate into ions that conduct electricity (Tolmie and Biggs, 2000). EC is measured in Siemens per unit length, with the standard unit of deci-Siemens per metre (dS/m). It is a convenient measure of salinity since many salts disassociate to the ionic form in water. Since water naturally has a very low EC, the charge is due to the presence of the salts (SalCon, 1997).

For soils, the common laboratory measurement methods are 1:5 soil water suspension, soil saturation extract and electrical conductivity of soil at measured or maximum field content.

The more common measure of electrical conductivity is the 1:5 soil water suspension ($EC_{1:5}$). It is a simple and quick method of estimating soil salinity. It can be undertaken in the field or the laboratory (SalCon, 1997). However, the measured water content is more dilute than field conditions (Shaw, 1999). This method tends to underestimate the EC of sandy soils compared with clay soils. Also, the presence of sparingly soluble salts cause over-estimation of salinity (SalCon, 1997).

The saturation extract electrical conductivity method (EC_{se}) is a useful laboratory method for relating plant response to salinity for a wide range of soil textures. It is a more meaningful measure than $EC_{1:5}$, since it is closer to field water content. (EC_{se} is the most dilute soil solution concentration likely to be encountered by plants). However, it is a tedious method. There are also problems with reproducibility of the saturation water content. It is more commonly predicted from $EC_{1:5}$ and soil properties (SalCon, 1997 and Shaw, 1999).

It is important to realise that soil salt measurement is not an accurate process. Errors are introduced because soil properties vary and because of differences between methods, including:

- The method used to determine saturated water content and hence the amount of water added to the soil for the saturation extract method.
- Disparities introduced through the use of different soil grinding machines.
- Differing periods of equilibration that affect the proportion of partially soluble salts, such as gypsum, that are dissolved. (The optimal period of equilibration will be a compromise between time needed for dissolving partially soluble salts, biological activity that enhances solubility and time for sedimentation of dissolved clay).
- Different soil solution extraction methods produce different results.

Many of these error sources vary further with soil properties (Shaw, 1999).

To relate EC measurements to plant salt tolerance data, soil leaching and soil behaviour, the data must be in the EC_{se} form. $EC_{1:5}$ conversion to EC_{se} is likely to produce a less accurate result than direct measurement of EC_{se}. However, because EC_{se} is an imprecise measure and a difficult technique, prediction of EC_{se} from $EC_{1:5}$ may be the most appropriate measure. (Shaw, 1999).

EC_{se} determined by analysis or calculated from $EC_{1:5}$ is the recommended measure of soil solution salinity. Laboratory methods for salt measurement should be undertaken according to Rayment and Higginson (1992).

A range of formulae are available for converting *$EC_{1:5}$* to *EC_{se}*. The SALF software available free from Department of Natural Resources and Mines (Queensland) includes a SALFCALC component that readily converts between EC methods at different salinities, based on soil properties (SalCon, 1997). However, a simple extrapolation from texture is given in Shaw (1999). This is presented as Figure 3, with clay content of soil based on the data presented in Table 40.

From SalCon (1997):

$EC_{se} = EC_{1:5}(500 + 6ADMC / SP)$

Where:

ADMC = air dry moisture content measured between 401C and 1001C expressed as kg/100 kg

SP = saturation percentage.

The conversion is not straightforward. To make an accurate estimate, both the ratio of water contents and the salt composition need to be known.

FIGURE 7 – RELATIONSHIP BETWEEN EC_{se} AND MEASURED $EC_{1:5}$

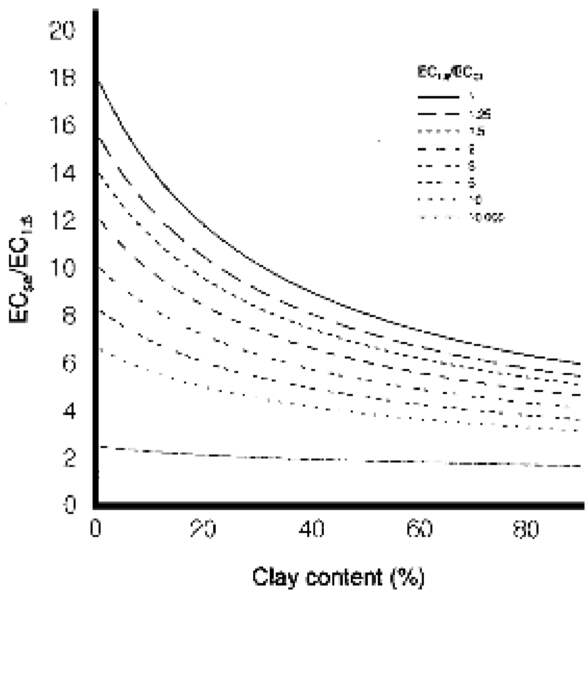

TABLE 62 – RELATIONSHIP BETWEEN TEXTURE CLASS, TEXTURE GRADES AND CLAY CONTENT

Texture Class	Texture Grades of McDonald & Isbell (1990)	Median Clay Content, approx. % (Shaw, 1994)
Sand	Sand	5
Loamy sand	Loamy sand, clayey sand	7
Sandy loam	Sandy loam	15
Silty loam	Loam, silty loam	25
Clay loam	Clay loam, silty clay loam	32
Light clay	Light clay, light medium clay	40
Medium clay	Medium clay	50
Heavy clay	Heavy clay	65

(Cited by Shaw 1999).

Because plants respond to salinity throughout the root zone, it is also useful to convert EC_{se} at a number of depths to a single value representing the entire root zone. Root zone salinity is commonly expressed as either the average root zone salinity or the water uptake weighted root zone salinity. Both methods require an estimate of the root depth of the particular plant being grown.

Average root zone salinity is the sum of the salinity measurements for a series of root zone depth increments divided by the number of root depth increments. It provides a conservative measure of plant response since plants respond to both atmospheric and soil conditions. However, it is a reasonable estimator of plant response to salinity (SalCon, 1997 and Shaw, 1999).

A more realistic predictor of plant response to salinity is w*ater uptake weighted root zone salinity*, which is based on the actual water uptake pattern of plants. With increasing depth, many Australian soils have increased salinity with reduced soil porosity, hydraulic conductivity and water storage capacity. Consequently, plant water uptake is not uniform throughout the root zone depth. Most water uptake is from the upper root zone depths. Rainfall and irrigation patterns also influence the water uptake pattern. Water uptake weighted root zone salinity probably better represents root zone salinity where subsoils are saline or where there are shallow water tables (and consequently saline surface soils). However, it may not be conservative enough to represent the plant response during dry periods when subsoil water is essential for plant survival. Average root zone salinity may provide a better estimate in these cases (Shaw, 1999).

Water uptake weighting patterns at 0.1 m increments are given for three common rooting depths in Table 63. Multiply the actual EC measurement at each depth by the appropriate weighting factor for the root zone depth of interest and sum the values to find the water uptake weighted root zone salinity (SalCon, 1997). Table 64 provides water uptake weighting pattern factors for standard survey depths and three common rooting depths.

TABLE 63 – WATER UPTAKE WEIGHTING PATTERN FACTORS (WUW) FOR 0.1 m DEPTH INCREMENTS FOR THREE COMMON ROOT ZONE DEPTHS

Soil increment (m)	Weighting factor for each 0.1 m increment where root zone depth is:			Analysed EC_{se} (dS/m)	Weighted EC_{se} (dS/m) (EC * weighting factor for 0.9 m deep soil)
	0.6 m	0.9 m	1.2 m		
0-0.1 m	0.35	0.27	0.23	0.4	0.10
0.1-0.2 m	0.18	0.14	0.12	0.4	0.06
0.2-0.3 m	0.15	0.11	0.10	0.4	0.05
0.3-0.4 m	0.13	0.10	0.08	0.5	0.05
0.4-0.5 m	0.11	0.09	0.07	0.7	0.06
0.5-0.6 m	0.08	0.08	0.07	1.1	0.09
0.6-0.7 m		0.08	0.07	1.9	0.15
0.7-0.8 m		0.07	0.06	3.2	0.22
0.8-0.9 m		0.06	0.06	4.2	0.25
0.9-1.0 m			0.06	average	wuw
1.0-1.1 m			0.05	root zone	- sum of
1.1-1.2 m			0.03	Mean	Values
SUM	1.0	1.0	1.0	=1.42	=1.03

(Shaw *et al.* 1987).

TABLE 64 – WATER UPTAKE PATTERN WEIGHTING FACTORS FOR STANDARD SURVEY DEPTHS AND THREE COMMON ROOTING DEPTHS

Soil increment (m)	Weighting factor for each 0.1 m increment where root zone depth is:		
	0.6 m	0.9 m	1.2 m
0-0.1 m	0.35	0.27	0.23
0.2-0.3 m	0.46	0.35	0.10
0.5-0.6 m	0.19	0.25	0.07
0.8-0.9 m	-	0.13	0.06
1.1-1.2 m	-	-	0.03
SUM	1.0	1.0	1.0

(based on Shaw *et al.* 1987).

For effluent reuse areas, and other irrigated land, the conversion of EC_{se} to water uptake weighted root zone salinity is recommended.

TDS

TDS is a measure of total dissolved solids on a mass per unit volume basis. It can be measured by evaporation or calculated. TDS by evaporation (TDS evap) is the weight of material remaining after the sample filtrate is evaporated and dried to a constant weight at a specified temperature. The calculation is total silica plus the sum of the cations and anions less ($HC0_3^-$ * 0.5083) expressed on a mass per volume basis. The calculation must involve the sum of at least Ca^{2+}, Mg^{2+}, Na^+, CO_3^{2-}, HCO_3^-, SO_4^{2-} and Cl^-. (NOTE: the bicarbonate correction allows for the conversion of HCO_3^- to CO_3 on evaporation. This measure approximates the results for TDS by evaporation) (SalCon 1997).

TDI

TDI is the sum of the analysed cations plus anions expressed on the basis of mass per volume. The ions considered must include at least Ca^{2+}, Mg^{2+}, Na^{+}, CO_3^{2-}, HCO_3^{-}, SO_4^{2-} and Cl^{-}. (This is also the measure of Total Soluble Salts (TSS)) (SalCon 1997).

The TDI method is inferior to TDS since it excludes total silica and does not account for the conversion of HCO_3^{-} to CO_3 on evaporation. Consequently, it is not recommended for use with intensive animal industries under the Load Based Licensing scheme.

Appendix D. SODICITY: ANALYSIS METHODS

Measuring Sodicity

The sodium adsorption ratio (SAR) is used to examine sodicity in liquids. It is a useful measure since there is a close relationship between SAR and the ESP of the soil. SAR is the amount of sodium relative to calcium and magnesium in a soil solution or water that approximates the exchangeable sodium percentage (ESP) of the soil (SalCon, 1997).

$SAR = [Na]/(0.5 * [Ca]+[Mg])^{0.5}$
Where concentrations are in meq/L (SalCon, 1997).

Soil ESP can be calculated from SAR using the relationship of USSL (1954) (cited by SalCon, 1997):

$ESP = (100 (-0.0126 + 0.01475 SAR)) / (1 + (-0.0.0126 + 0.01475 SAR)$

Because divalent cations are preferentially adsorbed onto clay exchange sites, the proportions of Ca^{2+}, Mg^{2+} and Na^{+} on the soil exchange do not match the proportions in the soil solution. The reverse equation can be use to derive SAR from ESP:

$SAR = 0.6906 ESP^{1.128}$
(This equation is applicable for ESP values of 0-50). (SalCon, 1997).

Residual alkali (RA) provides a measure of the effect of effluent irrigation on soil properties. RA measured the excess of sodium bicarbonate and carbonate ions in the water over calcium and magnesium ions. When these salts combine with calcium and magnesium in the soil solution, they are removed by precipitation leaving an excess of sodium ions in the soil. However, RA on its own in not a useful measure of sodicity hazard since water may have a high SAR but a low RA (SalCon 1997).

An indicator for soil sodicity is the Exchangeable Sodium Percentage (ESP), which is the amount of sodium ions adsorbed by clay particles as a percentage of total Cation Exchange Capacity (CEC) (SalCon, 1997). The calculation for exchangeable sodium percentage (ESP) is:

ESP (%) = (Exchangeable Sodium (meq/100 g)/CEC (meq/100 g)) * 100

Cation Exchange Capacity (CEC) is the total amount of cations on the surface layer of clay materials that are readily exchanged with other cations available in solution, expressed as millequivalents per 100 grams of dry clay (meq./100 g) (SalCon, 1997). The dominant exchangeable cations in most soils are: Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}. Exchangeable cations present in smaller percentages can include: NH_4^{+}, Cu^{2+}, Co^{2+} and Zn^{2+}. Aluminium, iron and hydrogen cations may also be present in acidic soils (Rengasamy and Churchman, 1999).

Effective CEC (meq./100 g) = (Exch. Ca) + (Exch. Mg) + (Exch. Na) + (Exch. Ca)
where all units are in meq./100 g (Rengasamy and Churchman, 1999).

Soil SAR can be measured. However, relationships been between ESP and SAR in soil solutions vary depending on both the method of extraction and soil properties. Hence, these should be applied with caution (Rengasamy and Churchman, 1999).

Although Australian soils are generally regarded as sodic if the ESP exceeds ~ 5%-6%, the actual ESP at which soils become sodic depends on a range of soil properties. The concentration of salts in solution is particularly important as a determinant of the expression of sodicity for a given soil (Rengasamy and Churchman, 1999). Simple field tests are available to assess the degree of turbidity produced when soil is gently mixed with distilled water. Sodic soils disperse, creating turbidity. Non-sodic soils do not (Rengasamy and Bourne, 1997).

Sodicity may be better defined by soil behaviour than by indices related to soil composition (Rengasamy and Churchman, 1999).

Analysis Methods

Analysis results for exchangeable cations and CEC in the soil vary depending on pH and ionic strength, along with other factors. Rayment and Higginson (1992) include methods for:

- Non-calcareous soils dominated by permanent charge - exchange with 1M NH_4Cl, pH 7.0
- Calcareous soils dominated by permanent charge - exchange with 1M NH_4Cl, pH 8.5
- Soils dominated by variable charge - exchange with 0.01M silver thiourea <u>or</u> compulsive exchange with $BaCl_2$ / NH_4Cl
- Determining exchange acidity – exchange with 1 M KCl <u>or</u> exchange with $BaCl_2$ in triethanolamine at pH 8.2 and acid titration of excess triethanolamine
- Removing soluble salts as pre-treatments – pre-treatment with a solvent e.g. aqueous ethanol or aqueous glycerol.

A study by Maheswaran and Peverill (1995) (cited by Rengasamy and Chuchman, 1999) revealed large variations in analysis results for exchangeable cation and CEC measurements made on the same samples by different Australian laboratories. Coefficients of variation ranged from 65-164% for individual cations and were 62% for CEC. The variation in results depended strongly on the analysis method.

Most Australian soil testing laboratories use ammonium acetate or barium chloride to displace exchangeable cations from air-dried and ground soil samples. The former method is suitable if soil pH is 5.0-7.5 and few salts are present. However, in the present of salts such as gypsum and lime, cations should be extracted using ammonium chloride buffered at a pH of 8.5 to prevent dissolving of lime. It is also important to determine the sulfate content of the leachate to correct for dissolved gypsum (McKenzie, 1990).

CEC and exchangeable cation should be determined using the methods in Rayment and Higginson (1992). Where possible the same laboratory and analysis method should be used for samples collected regularly from monitoring sites.

Appendix E. Salinity and Sodicity Tolerance of Crops

Table 65 – Salinity Threshold, Productivity Decrease and Soil Salinity at 90% Yield for Various Crops and Pastures

Common name	Salinity threshold (EC_{se})	Productivity decrease per dS/m increase (%)	Soil salinity Ec_{se} at 90 % yield
Grains			
Barley, grain	8.0	5.0	10
Corn, grain, sweet	1.7	12.0	2.5
Cotton	7.7	5.2	9.6
Cowpea (seed)	1.6	9.0	2.7
Cowpea, Caloona	2.0	10.8	2.9
Flax/Linseed	1.7	12.0	2.5
Oats	5.0	20.0	5.5
Peanut	3.2	29.4	3.5
Phasey bean, Murray	0.8	7.9	2.1
Rice, paddy	3.0	12.2	3.8
Safflower	6.5		
Sorghum	6.8	15.9	7.4
Sorghum, crooble	8.3	11.2	9.2
Soybean	5.0	20.0	5.5
Sugarcane	1.7	5.9	3.4
Sunflower	5.5	25.0	5.9
Wheat	6.0	7.1	7.4
Wheat, durum	5.7	5.4	7.6
Fruits			
Apple	1.0	18.0	1.6
Apricot	1.6	23.0	2
Avocado	1.3	21.0	1.8
Grape	1.5	9.5	2.6
Grapefruit	1.8	16.1	2.4
Lemon	1.0		
Olive	4.0		
Orange	1.7	15.9	2.3
Peach	3.2	18.8	3.7
Pear	1.0		
Plum	1.5	18.2	2
Pastures			
Barley, forage	6.0	7.0	7.4
Barley, hay	6.0	7.1	7.4
Barrel medic, Cyprus	3.0	14.6	3.7
Barrel medic, Jemalong	1.0	7.7	2.3

Buffel grass, Gayndah	5.5	10.3	6.5
Buffel grass, Nunbank	6.0	6.8	7.5
Clover, alsike, ladino, red	1.5	12.0	2.3
Clover, berseem	2.0	10.3	3
Clover, berseem (USA)	1.5	5.8	3.2
Clover, rose (Kondinin)	1.0	8.9	2.1
Clover, strawberry (Palestine)	1.6	10.3	2.6
Clover, white (New Zealand)	1.0	9.6	2
Clover, white (Safari)	1.5	12.1	2.3
Corn, forage	1.8	7.4	3
Couch grass	6.9	6.4	3.2
Cowpea (vegetative)	1.3	14.3	2.1
Desmodium, green leaf	2.1	14.9	2.6
Desmodium, silverleaf	1.0	22.7	2
Dodonea	1.0	7.8	2.3
Dolichos Rongai	1.0	15.6	1.6
Fescue	3.9	5.3	5.8
Glycine tinaroo	1.8	9.9	2.8
Green panic, Petri	3.0	6.9	4.4
Kikuku grass, Whittet	3.0	3.0	6.3
Leichardt	3.0	15.6	3.6
Lotonis, Miles	1.0	12.2	1.8
Lovegrass	2.0	8.5	3.2
Lucerne, Hunter River	2.0	6.0	3.7
Lucerne, Hunter River (temperate)	1.5	6.9	2.9
Lucerne (USA)	2.0	7.3	3.4
Meadow foxtail	1.5	9.7	2.5
Orchard grass	1.5	6.2	3.1
Pangola grass	2.0	4.0	4.5
Paspalum	1.8	9.0	2.9
Phalaris	4.2		
Rhodes grass, Pioneer	7.0	3.2	10.1
Sesbania	2.3	7.0	3.7
Setaria, Nandi	2.4	12.2	3.2
Siratro	2.0	7.9	3.3
Snail medic	1.5	12.9	2.3
Strand medic	1.5	11.6	2.4
Sudan grass	2.8	4.3	5.1
Townsville stylo	2.4	20.4	2.9
Trefoil, big	3.0	11.1	3.9
Trefoil, birdsfoot	5.0	10.0	6
Urochloa	8.5	12.4	9.3
Wheatgrass, crested	3.5	4.0	6
Wheatgrass, fairway	7.5	6.9	8.9
Wheatgrass, tall	7.5	4.2	9.9

Reprinted from SalCon (1997)

Appendix F. COLLECTION, STORAGE, HANDLING & TREATMENT OF SAMPLES

Surface Water - Quality

Before undertaking any water sampling, you should plan how this will be undertaken.

1. Decide on the sampling locations and the sampling frequency or triggers.
2. Select a suitable laboratory.
3. Identify couriers that can transport the samples to the laboratory (if needed).
4. Assemble the sampling equipment.
5. Clearly understand the sampling procedures.
6. Know the monitoring parameters.

Many regulatory agencies have their own water quality monitoring guidelines. Advice should be sought from the relevant agency before planning sampling and monitoring procedures. In the absence of specific advice from the applicable agency, the following guidelines may be used.

<u>Sampling Location</u>

If monitoring is a licence condition, the licence may specify sampling locations. If sampling locations are not detailed on your licence conditions, identify suitable sites that you can locate and access each time monitoring is required. Discuss selected sampling locations with the licensing authorities before sampling to ensure that the results will be acceptable.

For stream monitoring, take samples upstream and downstream of the area of interest. A sample should be taken immediately upstream and approximately 100 m downstream of an area of interest. The downstream sample should be taken some distance downstream of the area of interest to allow for mixing of any effluent with the stream water. However, if the distance between sampling points is too great, inflows from other sources may affect the analysis results. If another watercourse enters the relevant stream between the two sampling points, you should also sample water from the secondary watercourse close to it's junction with the watercourse of interest.

Contaminants within a terminal pond may disperse slowly. It is therefore appropriate to sample close to the entry point of runoff into the pond.

<u>Water Quality Monitoring Interval</u>

Water quality monitoring may be undertaken at a set interval (e.g. quarterly, six monthly or annually) or may be triggered by specific events (e.g. an overtopping effluent pond). Water quality varies with time of day, flow rate and recent weather conditions. Note these factors at the time of sampling.

If a spill to a watercourse is the trigger for sampling, you should sample during the spill. You should also sample the spilling effluent at this time.

Select a Laboratory

National Association of Testing Authorities, Australia (NATA) accredited laboratories are preferred for sample analysis. Check that the laboratory is NATA-accredited (or equivalent) for the analyses needed. Analysis methods vary between laboratories, which may affect results.

Select a Courier (if needed)

If you cannot take samples directly to the laboratory yourself, identify a courier that can transport the samples to the laboratory within the required time frame between collection and analysis.

Samples should arrive at the laboratory within two days of sampling and must be kept on ice over this whole time period. If this is not possible, you may need to freeze your samples (consult the laboratory). You should schedule sampling to coincide with courier dispatch to minimise the amount of time between sampling and analysis. *Ideally, sampling should occur on a Monday or Tuesday so that samples arrive at the laboratory and are promptly analysed rather than having to sit over a weekend.*

Assemble Sampling Equipment

The sampling equipment may include:

1. Appropriate sample containers and preservatives. Most laboratories will supply suitable sample containers, as well as any necessary preservatives. Water quality sampling manuals can also be consulted to determine sample container sizes and required preservatives. Obtaining sample containers from the laboratory reduces the chance of sample contamination and ensures that the sample size is adequate.

2. A sampling rod. A rod with a large clamp for holding the sampling container allows greater reach when sampling. Otherwise, you can wade to collect the sample. The sample should be taken from upstream of your feet to ensure that disturbed sediment is not collected.

3. A bucket that has been washed several times with clean water and then rinsed several times with the water to be sampled.

4. Cheap, styrofoam eskies.

5. Plenty of crushed ice to pack around the samples in the eskies.

6. Waterproof pen to mark sample bottles.

7. Waterproof tape to seal eskies.

8. Personal protective clothing. For example, waterproof boots if wading.

9. Analysis request forms. Most laboratories have their own analysis request forms and prefer these to accompany samples. Some of the details on the forms can be completed prior to sampling. (e.g. name, sampling location and analysis parameters). However, some details can only be completed at sampling (e.g. time of

sampling). If analysis request forms are not provided, you will need to make up your own.

10. Envelope that analysis request forms will fit in.

11. Pen to complete analysis request form.

<u>Collect and Dispatch Samples</u>

The following are suggested sampling procedures:

1. Assemble the sample containers and the sample preservatives.

2. With a waterproof pen, label the sample containers with the enterprise name, your telephone number, a unique sample number (new numbers should be used at each sampling), the sampling location (e.g. Deep Creek upstream of effluent irrigation area) and the date of sampling. Label the container instead of the lid, as lids can get mixed up in the laboratory.

3. Complete as many details of the analysis request forms as possible. This should include: contact details, sample numbers (matching those recorded on the sample bottles), sampling location, sampling date and analysis parameters.

4. Fill eskies with ice.

5. If you need to wade into a watercourse, first satisfy yourself that it is safe to enter. Hidden obstacles and rapid flowing water pose significant risks, particularly if alone.

6. Collect samples directly into sample containers. Either take a grab sample or a composite sample. A grab sample is one taken by quickly filling sample containers. A composite sample comprises several grab samples collected over several minutes. Composite samples comprising five grab samples should be collected if there is little movement in the watercourse or for dam samples. Stream samples should be collected midstream, clear of bank edges and other potential contaminant sources. If sampling from a terminal pond, take the sample away from the edge of the dam.

7. Remove the sample bottle lid, taking care not to touch the inside of the lid or bottle. Face the mouth of the bottle downwards and plunge into the water. Turn the bottle to a horizontal position facing the current preferably 0.2 m below the water surface (this avoids sampling surface scum). If necessary, create a current by dragging the bottle away from yourself. Remove the bottle as soon as it completely fills. If you are taking a composite sample, you should collect five samples over a period of a few minutes and thoroughly mix these in a clean plastic bucket before pouring the mixed water into a sample bottle. Add any required preservative and replace the lid.

8. Immediately place the sample in an esky, pack crushed ice completely around it and replace the esky lid. *Do not put effluent samples in the same esky as surface water samples.* Store the esky in the shade.

9. If samples will take longer than 48 hours to get to the laboratory, they should be frozen. Do not completely fill the sample bottle if you intend to freeze the sample.

10. When all other surface water or groundwater samples have been added to the esky, seal it with the waterproof tape.

11. Thoroughly wash your hands.

12. Complete the analysis request forms and photocopy for your own records (if you have access to a photocopier or fax machine). Place the original forms in an envelope. Clearly address the envelope to the laboratory and add their phone number. In smaller writing, put your own address and phone number on the envelope as "sender". Firmly tape the envelope to the top of the esky. Store the esky in the shade.

13. Deliver the samples or arrange for courier delivery.

14. Contact the laboratory to confirm that the samples were received within 48 hours of sampling.

Recording

At each water quality sampling, record:

1. Location and name of sampling site. The sampling location must be clearly identified so that you can return to the same site for future sampling

2. Date and time of day that each sampling occurs. Water quality varies over time.

3. Flow rate (in watercourses) or approximate depth of water in terminal ponds and weather conditions at the time of sampling. Water quality varies with flow rate.

4. Weather conditions at the time of each sampling. This may influence water quality.

5. Method of sampling. For instance, grab sample or composite sample.

6. Name of sampler.

7. Time between sampling and dispatch of sample to laboratory.

8. Method of preserving samples (e.g. sample immediately put on ice in esky).

9. Time samples dispatched to laboratory.

10. Analysis parameters requested. (Preferably keep a copy of the original analysis request forms).

Remember to keep the original copy of any laboratory analysis reports.

Groundwater Quality

Refer to surface water quality section.

Sampling Location

If monitoring is a licence condition, the licence may specify sampling bore or piezometer locations. A piezometer is a non-pumping well generally of small diameter with a short screen through which groundwater can enter. If sampling locations are not detailed on your licence conditions, or if you are undertaking voluntary monitoring, you will need to identify or install suitable monitoring bores or piezometers. These must be installed correctly. Depth and casing are particularly important. Monitoring bores or piezometers may also need to be registered prior to construction. Consult the piggery regulatory agency in your state.

As groundwater may move extremely slowly, bores or piezometers should be located in close proximity, and downstream, of the area for monitoring. It is also advisable to locate a bore or piezometer above the area of interest for comparison purposes. Ensure that both bores are tapping into the same aquifer. A network of bores will provide better information than a single monitoring bore plus background bore. However, budgetary constraints will often preclude the installation of several bores.

Groundwater Quality Monitoring Interval

Groundwater quality monitoring is usually undertaken at a set interval (e.g. quarterly, six monthly or annually).

Select a Laboratory

Refer to surface water quality section.

Select a Courier (if needed)

Refer to surface water quality section.

Assemble Sampling Equipment

Refer to surface water quality section.

Also:

- A sampling bailer or pump. You will need to use a bailer or pump to draw water from the monitoring bores. If you are using a bailer, wash it thoroughly with water before use. A bailer is a time consuming method for sampling groundwater. It is also impractical for deep bores. A pump is convenient to use and allows for samples to be quickly collected.

- A tape measure to determine depth to groundwater.

Collect and Dispatch Samples

1-4. See steps 1-4 of “Collect and Dispatch Samples” of surface water quality section.

5. Measure the depth to groundwater. Pump several bore volumes from the casing to ensure that you are not sampling stagnant water. Sometimes this will take quite a while.

   Bore volume (L) = ((3.14/1000) * (radius m)^2) * water depth (m)

   Collecting grab samples of standing water may provide misleading results since the groundwater quality may be stratified and interactions between the bore casing and atmosphere of the water may influence water quality properties. If it is not possible to purge the bore prior to sampling, the sampling process should disturb the water within the bore as little as possible. For shallow piezometers, it may be appropriate to empty the piezometer one to two days prior to sampling and then to allow it to refill.

6. Allow bore to recharge with groundwater. Measure the depth to groundwater. Collect a grab sample using a bailer or pump.

7. Remove the sample bottle lid, taking care not to touch the inside of the lid or bottle. Fill the bottle directly from the bailer or pump. Remove the bottle from the flow as soon as it completely fills. Add any required preservative and replace the lid.

8.-14. See steps 8-14 of "Collect and Dispatch Samples" of surface water quality section.

<u>Recording</u>

Refer to surface water quality section.

Also:

- Name and location of bore or piezometer.
- Depth to groundwater.

Effluent, Solid By-Products, Plants and Soils Quality Monitoring

Refer to steps 1-6 in the surface water quality section.

<u>Sampling Location</u>

Effluent

Effluent should be sampled from the sampling stopcock, priming plug or main outlet of the effluent irrigation pump. If this is not possible, collect the sample from the pond from which irrigation water will be drawn.

Solid By-Products

For each type of solid by-product a separate sample is needed (e.g. screenings, sludge, spent bedding and each type of compost). If screenings are spread fresh, then a fresh sample should be collected. If screenings are composted before spreading, then a composted sample should be collected.

Soils

For soils each sampling location should represent a particular type of soil and general land use (including land use and effluent or solids spreading rates).

The following steps will help you decide how many sampling locations are needed:

- Divide each area used for effluent irrigation or solids spreading according to soil types. Dig some holes and compare the soils of each hole. (Recording information as you go is important!).
- Divide each area on the basis of land use as sustainable spreading rates vary widely depending on whether the land is grazed or used to grow a crop. Areas with different land uses should be monitored separately. However, it is not necessary to provide a monitoring plot in each separate paddock if there are similar land uses between paddocks with the same soil type.
- Divide each area on the basis of by-product type (e.g. effluent, screenings, sludge, spent litter or compost) and application rate.

  For instance, there might be two major soil types on your farm. If both soil types are used for growing cereal crops and for effluent irrigation, but at two different rates, you have four different soil type/land use combinations (soil 1 low rate, soil 1 high rate, soil 2 low rate, soil 2 high rate). Similarly, if there is one soil type, but two different land uses (e.g. cereal crops V grazing), you will have two soil type/land use combinations (soil 1 land use 1, soil 1 land use 2).
- Identify a 20 m diameter sampling plot for each soil type, by-product and land use combination. This area should be representative of the area most at risk. For instance, if you have two areas of land with similar soils and land uses but different effluent application rates, you may monitor only the area with the highest effluent application rate. This area should also be free from stumps, atypical rockiness, tracks, animal camps and other unusual features.
- For each soil type to be monitored, you should also locate a 20 m diameter background monitoring plot on an area that has not been used for effluent irrigation, solids spreading or conventional fertiliser spreading. This will be used to compare with monitoring plot data from the effluent and solids spreading areas. It is recognised that it is not always easy to find a suitable background plot.
- Mark the location of the plots on your Property Map so that you can come back to same area in subsequent years. (Keep using these sites from year to year).

Plant Samples

Any plant samples taken should be representative of the material being harvested. For a grain crop, collect samples from the field bin (or similar). For a baled crop, collect samples of hay. For a silage crop, collect samples of freshly cut material from several bales or bins.

<u>Select a Laboratory</u>

Refer to surface water quality section.

<u>Select a Courier</u> (if needed).

Refer to surface water quality section.

<u>Assemble Sampling Equipment</u>

Effluent

See “Assemble Sampling Equipment” in surface water quality section.

Solid By-Products

See “Assemble Sampling Equipment” in surface water quality section.

Sampling containers will either be wide-mouthed sampling bottles or plastic bags. Bottles may better suit high moisture sludge. It is recommended that you obtain these from the chosen laboratory. Bags will suit drier products.

Also:

- A shovel.
- A small garden trowel.

Soils

See “Assemble Sampling Equipment” in surface water quality section.

Also:

- Soil auger or hydraulic soil sampling rig (these can be hired).
- Plastic sample bags. Most laboratories will supply suitable sample bags.
- Ruler or tape measure.
- Hand trowel.
- Plastic sheet.
- A bucket that has been washed several times with clean water.

Plants

The sampling equipment may include:

- Large paper sample bags. Most laboratories will supply suitable sample bags. Brown paper bags will do.
- Disposable gloves.
- Clean sampling cup.

- Clean bucket.
- Waterproof pen to mark sample bags.
- Analysis request forms.
- Envelope that analysis request forms will fit in.
- Pen to complete analysis request form.
- Box to put samples in.

<u>Collect and Dispatch Samples</u>

Effluent

1-4 See steps 1-4 of the Collect and Dispatch Samples section of surface water quality section.

5. Put on disposable gloves if sampling effluent. Avoid splashing eyes with effluent or sample preservatives. Do not inhale aerosols from the effluent being sampled or the preservatives. Do not eat, drink or smoke and carry out standard hygiene practices.

6. If sampling from a pump: Start the pump and allow it to run for at least 10 minutes prior to collecting samples. While you are waiting, rinse the bucket several times with the effluent from the pump. Remove the sample bottle lid taking care not to touch the inside of the lid or bottle. Sample the effluent by collecting a cup of effluent in the sampling bottle every three to four minutes and adding each of these to the bucket until it is half full (10-15 samples). Thoroughly mix the effluent by swirling the bucket. Fill the sample bottle from the composite sample. Add any required preservative and replace the lid.

7. If sampling from the pond: Rinse the bucket several times with pond effluent. Remove the sample bottle lid taking care not to touch the inside of the lid or bottle. Sample effluent by facing the mouth of the bottle downwards and plunging it into the water. Turn the bottle to a horizontal position 0.2 m below the water surface (this avoids sampling surface scum). Create a current by dragging the bottle away from yourself. Remove the bottle as soon as you have filled the container and pour the effluent into the bucket. Repeat this procedure four times sampling from a different spot in the pond each time. When you have collected five samples, thoroughly mix these before pouring the composite sample into a sample bottle. Add any required preservative and replace the lid.

8-14 See steps 8-14 of the Collect and Dispatch Samples section of surface water quality section.

Solid By-Products Quality

1-4 See steps 1-4 of the Collect and Dispatch Samples section of surface water quality section.

5. Put on disposable gloves and dust mask (if sampling dusty products). When sampling, do not eat, drink or smoke and carry out standard hygiene practices.

6. If sampling from a pump e.g. sludge: Start the pump and allow it to run for at least 10 minutes prior to collecting samples. Remove the sample bottle lid taking care not to touch the inside of the lid or bottle. Sample the sludge by collecting a cup in the sampling bottle every three to four minutes and adding each of these to the bucket until it is half full (10-15 samples). Thoroughly mix the sludge by swirling the bucket. Fill the sample bottle from the composite sample. Add any required preservative and replace the lid.

7. If sampling from a stockpile (screenings, spent litter, compost): Use a clean shovel to collect 25 samples of solids (sample size should be about a cup). As you collect each sample, place in the bucket and thoroughly mix with the garden trowel. Place about four cups of the mixed sample into a bottle or bag and seal. Put the bag or bottle inside another bag and seal well.

8-14 See steps 8-14 of the Collect and Dispatch Samples section of surface water quality section.

Soils

1-4 See steps 1-4 of the Collect and Dispatch Samples section of surface water quality section.
When labelling the sample bags, remember to include the sampling depth (e.g. 0-0.1 m).

5. From random locations within each 20 m diameter sampling plot, collect 25 equal-sized samples of soil to a depth of 0.1 m (10 cm). As you go, record a description of the soil sampled. Combine all of the samples in the bucket and thoroughly mix using a hand trowel. Remove rock fragments exceeding 2 cm diameter and large roots. Break up large clods.

6. Pour the mixed composite sample into a cone on the plastic sheet. Divide the cone into four quarters. Discard three and thoroughly mix the remaining quarter. Repeat the procedure with the remaining quarter until the sample size is small enough to fill the sample bag (generally about 0.4-0.5 kg or 1 lb). Fill the sample bag and immediately place it in an esky.

7. From random locations within each 20 m diameter sampling plot, drill at least five holes to collect subsoil samples. (Drilling more holes provides a more reliable sample. Eight holes are preferred). As you go, record a description of the soil encountered.

   Samples should be collected from the 0.2-0.3 m (20-30 cm) and 0.5-0.6 m (50-60 cm) depths. If the base of the root zone is below 0.6 m, it is also useful to collect a deeper sample (1.5 – 2.0 m). Combine all of the samples from the same depth in the bucket and thoroughly mix using a hand trowel. Remove rock fragments exceeding 2 cm diameter and large roots. Break up large clods. Use the same mixing and sub-sampling procedure as for the 0-0.1 m sample to obtain a 0.4-0.5 kg sample. Place the sample in the esky.

Either a bulked sample representative of the entire crop or pasture root depth, or alternatively, a number of samples at different intervals, could be sampled and analysed to determine the phosphorus sorption isotherm.

Never bulk (mix) soils of two different types.

Never mix soil layers (profiles) that are clearly different from each other.

Never bulk in depths greater than 0.3 m.

8-14 See steps 8-14 of the Collect and Dispatch Samples section of surface water quality section.

It is useful to take note of any unusual changes in the soils and plants of the effluent irrigation areas. These include:

- Free water on the soil surface may indicate waterlogging. Other signs include reduced plant growth, growth of weeds (dock, nutgrass) and drooping foliage with pale leaves.
- A surplus of nitrogen may be indicated by invasion of an area with nettles or fat hen.)
- Yellow or browned off vegetation is indicative of toxic nutrient levels or nutrient deficiencies.
- Bare patches in paddocks. These may indicate poor germination due to excess salinity. White crusting on soil surface in dry times may indicate evaporation from a shallow saline water table.
- Areas in effluent-irrigated paddocks that are consistently bare of vegetation may indicate too much salinity.

Plants

1-3 See steps 1-3 of the Collect and Dispatch Samples section of surface water quality section.

4. Collect the sample. If possible, this should occur between 8 AM and 11 AM.

5. For grain, it is suggested that at least five samples be collected from the field bin (or similar). These should be placed in the bucket and thoroughly mixed with gloved hands. A sub-sample should then be used to fill the sample bag.

6. For hay or cut forage, collect five sub-samples, thoroughly mix together in a bucket using gloved hands and sub-sample to fill the sample bag.

7. Leave the tops of the paper bags open to allow excess moisture to escape.

8. Put the bags in a box and leave in the shade or a cool place. Do not seal plant or grain samples in plastic bags or leave samples in the sun as they will sweat and degrade.

9. When the samples are ready for delivery, fold the tops of the bags over and fasten with staples or sticky tape. Place back in the box.

10. Complete the analysis request form and photocopy for your own records (if you have access to a photocopier or fax machine). Place the original forms in an envelope. Clearly address the envelope to the laboratory and add their phone number. Also add you own address and phone number marked “sender”. Firmly table the envelope to the top of the box. Store the box in the shade.

11. Deliver the samples or arrange for courier delivery.

12. Contact the laboratory to confirm that the samples were received within 48 hours of sampling.

Quantity of Effluent Irrigated

Methods for measuring the quantity of effluent irrigated vary depending on the enterprise. A flow meter can accurately measure the effluent flow rate. In-line flow meters should be a non-corrosive type. Alternatively, non-contact ultra-sonic, Doppler, and non-contact magnetic flow meters that clamp to the outside of the pipe are available although they may be too expensive.

A depth gauge in the pond, used with a storage capacity curve, can provide an estimate of the irrigation rate when large volumes are irrigated at a time. The curve shows the volume of effluent in the pond when filled to any depth. The change in depth from the start to the finish of the irrigation should be measured.

For a single hand-shift type sprinkler, the pumping rate can be estimated from the time taken to fill a container of known volume. The flow rate must be measured from the irrigation nozzle. It can be very difficult to measure effluent volumes this way. A plastic hose fitted over the nozzle and a 10 L bucket will help. For a sprayline, the outflow from at least three nozzles should be measured. Both sides of double-sided nozzles should be measured. As long as there are not too many pipe-join leaks, this method will give a good estimation.

If effluent is pumped from a tank or sump of known capacity, daily or weekly irrigation volumes may be estimated from the sump or tank volume and the emptying frequency.

If bulk tankers are used to spread effluent, tanker volume and emptying frequency provide a good estimate of the irrigation rate.

The quantity of effluent irrigated, and the paddock involved, should be recorded each time irrigation occurs.

Quantity of Solid By-Products Spread

If a tanker of a known volume (L or m^3) is used to spread the wet solids, it should be possible to estimate the number of loads per hectare quite easily. If a manure or fertiliser spreader is used, the spreading rate may be calculated from the volume of the storage hopper, the area of land for spreading and the bulk density of the solids (as per tanker method). Alternatively, you can determine the mass of the solids by weighing the truck or spreader filled with solids then subtracting the net weight of the truck or spreader.

The quantity of solids spread, and the paddock involved, should be recorded each time spreading occurs.

Yield of Plants or Liveweight Gain

It is generally adequate to estimate the nutrients removed from an area by yields and textbook nutrient concentrations of plants.

Measure yield of plants harvested by weighing or by estimating weight from the number of truck-loads removed. For a crop, the yield from an area should be recorded and a yield per hectare calculated (divide the total yield for the paddock (t) by the area of the paddock (ha)). The yield should then be converted to a dry matter yield. As a guide, grain crops have a dry matter content of about 88% and hay has a dry matter content of about 90%. Fresh harvested forage crops vary more.

If you harvest 4 t/ha of barley, the dry matter yield is about 3.5 t/ha (4 t/ha X 88/100). From Table 28, a 4 t/ha winter cereal crop removes about 80 kg N/ha and 12 kg P/ha. Hence, the 3.5 t/ha crop will remove about 70 kg N/ha and 10.5 kg P/ha (i.e. 80 kg N/ha X (3.5t/4t); 12 kg P/ha X (3.5 t/4t)).

Laboratory determination of the dry matter and plant tissue analysis can more accurately determine the nitrogen and phosphorus concentration of the harvested material. *This should only be required in border-line cases, for example where removing sufficient nutrients relies on luxury uptake.*

Monitoring Interval

Effluent and Solid By-Products

This should be based on the level of environmental risk. If monitoring results for the quality of the effluent or solid by-products over several years indicates similar results, the level of monitoring should be reduced from every year to say every three years.

Soils

This should be based on the level of environmental risk.

Sampling should occur at the end of a cropping cycle or at a time when nutrients are most vulnerable to leaching (before the onset of the wet season).

Plants

For most enterprises, analysis of plant composition should not be required. At a maximum, this should be once per crop (at harvest).

<u>Recording</u>

Quantity of Effluent Irrigated and Solid By-Products Spread

Each time effluent is irrigated or solids are spread on-farm, record the date, the paddock involved and the quantity of effluent (m^3 or ML) or solids (m^3, ML or t) involved. Also calculate the application rate (m^3/ha, ML/ha or t/ha).

If effluent or solids are removed off-site, record the date, the volume of material involved, the type of material involved, the recipients name and the proposed use (e.g. where the material will be irrigated or spread, the land use of the area involved and the application rate).

Effluent and Solid By-Products Quality

It is suggested that original copies of effluent and solid by-product analyses be kept for at least five years or as required by your licensing conditions. Use the analysis results to calculate appropriate irrigation or spreading rates depending on possible land uses.

If effluent or solid by-products are reused off-site, provide recipients with a copy of the analyses each time these products are analysed. Use the analyses to calculate appropriate irrigation or spreading rates depending on preferred land uses. Advise by-product recipients of the appropriate irrigation or spreading rates.

Soil Properties

Original copies of soil analyses should be kept indefinitely along with records of sampling locations and land use. This assists with long-term farm management.

Production from Land Area

Each time crops are harvested from effluent irrigation or solid by-products spreading areas record the yield harvested. Calculate the dry matter yield and the approximate nitrogen and phosphorus removal rates.

Appendix G. EXAMPLE RISK ASSESSMENT - PIGGERY

Description of Development

Capacity: 5150 SPU

Property size: 213.7 ha

Solid manure: All sold off-site to neighboring landholders

Piggery Site Description: The site comprises floodplains with average slopes of 0-0.2%. The plains comprise clay-loam to clay soils characterised by moderately deep gilgai. Table 66 shows a recent analysis of the soils of the effluent irrigation area.

Irrigation area description: The effluent irrigation area is 75 ha. Soils are typically brown, loose, light clays over light brownish-grey heavy clays over greyish-brown heavy clays. Soil sampling revealed that the soil depth is at least 60 cm. However, from experimental groundwater drilling it is known that the clays extend at least several metres.

TABLE 66 - ANALYSIS OF SOIL IN THE EFFLUENT IRRIGATION AREA.

Analysis Parameter		Concentrations		
Parameter	Units	0.0- 0.1m	0.2-0.3m	0.5- 0.6m
NO3-N	mg/kg	9	6	6
Total N	mg/kg	1020	-	-
P (Colwell)	mg/kg	40	-	-
K	mg/kg	520	240	200
Ca	mg/kg	1200	1800	1900
Mg	mg/kg	820	890	970
Al	mg/kg	28	21	23
Exch Na	meq/100 g	0.5	1.6	2.3
Exch K	meq/100 g	1.3	0.6	0.5
Exch Ca	meq/100 g	6.0	9.0	9.5
Exch Mg	meq/100 g	6.8	7.4	8.1
Exch Al	meq/100 g	0.3	0.2	0.3
EC	dS/m	0.06	0.11	0.13
Na	mg/kg	120	360	530
CEC	meq/100	15	18.8	20.7
pH	-	5.3	5.3	5.4
Organic Carbon	%	0.08	-	-

"-" indicates parameter not measured

P storage: Based on a 20-year reuse life, the site can safely store 98 kg/ha/yr of P. This was calculated from the P sorption isotherm, using a P solution concentration of 0.5 mg P/L.

Surface water: Because the property is quite flat, there are no distinct drainage lines. The closest creek is located about 2 km to the south of the property. The property is within the catchment of this creek.

Groundwater: There is no significant groundwater in the area surrounding the piggery. Clay layers at least several metres deep underlie the soils of the site.

Effluent storage: An anaerobic pond and a wet weather pond in series underlie the soils of the site. The ponds are adequately sized to limit overtopping to less than 1 in 10 years on average.

Effluent irrigation: Pond concentrations from the wet weather/irrigation pond have been measured and are: 1370 mg N/L, 150 mg P/L and 550 mg TDS/L. Some 10 ML of effluent needs to be irrigated annually to prevent pond overtopping, thus the amount of N and P irrigated annually is 13.7 t and 1.5 t respectively. The amount of salt irrigated annually as TDS is 5.5 t. Effluent is applied with a low-pressure hand-shift sprinkler.

Crop production: For the district forage sorghum crops typically yield 12 t DM/ha/yr, with N removal of 264 kg/ha/yr and P removal of 36 kg/ha/yr.

Risk Assessment

Risk Assessment: Nutrients in Manure and Effluent

Low Risk	The quantity of effluent and solids reused is measured and the quality of effluent and solids reused is regularly measured (at least annually, more frequently if required to ensure sound management of nutrients). OR You have developed a mass balance of nutrient production from your piggery or cattle feedlot using accepted design tools, such as PigBal, BeefBal or MEDLI using conservative figures. (There can be a great variation in nutrient predictions from mass balance models).
High Risk	You have never measured, but only estimated the mass of nutrients applied using “text-book” values, such as those provided in Section 6.3.12 (piggeries) and 6.6.9 (feedlots) of the Resource Manual.

A risk weighting of 1 or 3 applies to the Nutrients and Manure criterion. A low risk attracts a risk weighting of “1” and high risk attracts a risk weighting of “3”.

Transfer these values to the Risk Assessment Matrices for soils, surface water and groundwater.

As a mass balance of nutrient effluent application has been conducted by knowing concentrations and volumes irrigated, thus the risk assessment of nutrients in manure and effluent is low, so a number of **1** can be applied.

Risk Assessing the Site

Transfer data from this assessment into the Design and Management Reuse Area Risk Assessment Summary table.

Size of Land Area

Knowledge of Size of Land Area

Low Risk — From farm or paddock maps, you accurately know the area (ha) of each effluent or manure reuse paddock under each management regime (e.g. soil properties, land use).

Medium Risk — You know the approximate area (ha) of each effluent or manure reuse paddock under each management regime.

High Risk — You do not know the area of the effluent or manure reuse paddocks.

The area of the effluent irrigation area is approximately known from aerial photography, so the risk weighting is medium (**2**).

Knowledge of Yields of Crops or Pastures Grown on Reuse Areas

Low Risk — For your property and soil type, you know typical yields for the pastures or crops grown on reuse areas.

Medium Risk — You know typical district yields for the pastures or crops grown on reuse areas.

High Risk — You do not know typical yields for the pastures or crops grown on reuse areas.

Typical district yields for forage sorghum are known, thus the risk weighting is medium (**2**).

Knowledge of Nutrients Applied to Reuse Areas

Low Risk — You have calculated the nitrogen (kg/ha/yr) and phosphorus (kg/ha/yr) loading rates to reuse areas from estimated nutrient production.

High Risk — You have not calculated the nitrogen (kg/ha/yr) and phosphorus (kg/ha/yr) loading rates to reuse areas.

The nitrogen and phosphorus to be applied has been calculated i.e. 13.7 t N/yr spread over 75 ha = 182 kg/ha/yr, 1.5 t P/yr spread over 75 ha = 20 kg/ha/yr. Thus the risk weighting is low (**1**).

Nitrogen Mass Balance for Reuse Areas

Low Risk	You have calculated that the net mass of nitrogen (kg/ha/yr) applied as effluent and / or solid by-products is exceeded by the mass of nitrogen (kg/ha/yr) that plant harvest should remove.
Medium Risk	You have calculated that the net mass of nitrogen applied (kg/ha/yr) as effluent or solid by-products is equal to the mass of nitrogen (kg/ha/yr) that plant harvest should remove.
High Risk	The net mass of nitrogen applied to reuse areas (kg/ha/yr) exceeds the mass removed or you do not know the net mass of nitrogen applied to the reuse area.

The expected nitrogen removal rate by the crop is known to be 264 kg/ha/yr. After nitrogen volatilisation losses during irrigation (say 20%), the net nitrogen application rate is 145.6 kg/ha/yr. Thus the risk weighting is low (**1**).

Phosphorus Mass Balance for Reuse Areas

Low Risk	You have calculated that the mass of phosphorus (kg/ha/yr) applied as effluent and / or solid by-products is exceeded by the mass of phosphorus (kg/ha/yr) that plant harvest should remove plus phosphorus storage calculated from a site-specific phosphorus sorption test.
Medium Risk	You have calculated that the net mass of phosphorus applied (kg/ha/yr) as effluent or solid by-products is equal to the mass of phosphorus (kg/ha/yr) that plant harvest should remove plus phosphorus storage calculated from a site-specific phosphorus sorption test or from generic phosphorus sorption data for similar soil types.
High Risk	The net mass of phosphorus applied to reuse areas (kg/ha/yr) exceeds the mass removed plus storage calculated from a site-specific phosphorus sorption test or from generic phosphorus sorption data for similar soil types or you do not know the mass of phosphorus applied to the reuse area.

Typical data for the district suggests that the phosphorus removal rate by the crop should be some 36 kg P/ha/yr. The safe P storage capacity is 98 kg/ha/yr from site-specific P sorption data. The amount of P applied in effluent is only 20 kg/ha/yr, thus the risk weighting is low (**1**).

Using Appropriate Effluent & Solid By-Product Application Methods

If you reuse effluent on-site, select the appropriate risk category for “Effluent Irrigation” based on the information presented below. If you reuse solid by-products on-site, select the appropriate risk category for “Solids Spreading” from the information presented below. If you reuse effluent and solids on the same area, select the risk weighting that is highest from either the “Effluent Irrigation” or “Solids Spreading” sections below (e.g. if you have a rating of *low* for effluent irrigation and a rating of *medium* for solids spreading, the overall risk weighting you choose for the area is *medium*).

The results then need to be transferred into the "Using Appropriate Effluent & Solid By-Product Application Methods" row of Table 30 and converted into a risk weighting. A separate copy of Table 30 needs to be developed for separate reuse areas (e.g. effluent areas V solid areas) or reuse areas posing different risks (e.g. one effluent reuse area might be low risk, another high risk).

Effluent Irrigation

Low Risk	You use a low-pressure, travelling spray or drip irrigation system <u>or</u> a low-pressure solid set spray or drip irrigation system <u>or</u> a well designed and maintained flood irrigation system that is not on sandy to sandy loam soil. The system also applies effluent evenly and at target rates.
High Risk	You use a hand-shift sprinkler or hose <u>or</u> a poorly designed or managed flood irrigation system (e.g. land has not been levelled or effluent is unshandied or surface soil is sandy to sandy loam).

Effluent irrigation will be applied with a low-pressure hand shift sprinkler, thus the risk weighting is high (**3**).

Solids Spreading

Low Risk	The spreading method used disperses solids evenly and at target rates.
Medium Risk	The spreading method used disperses solids fairly evenly and within 20% of target rates.
High Risk	The spreading method used disperses solids unevenly or at uncontrolled rates (not within 20% of target rates).

Table 67 is a template for summarising the design and management risk weightings for each design and management criterion. To complete the table, insert a risk weighting of 1, 2 or 3 against each criterion. A low risk attracts a risk weighting of "1", medium risk attracts a risk weighting of "2" and high risk attracts a risk weighting of "3". These numbers are transferred to Table 54, Table 72 and Table 73.

All solid manure will be sold off-farm.

TABLE 67 - DESIGN AND MANAGEMENT REUSE AREA RISK ASSESSMENT SUMMARY

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)
Size of land area	**2**
Application methods	**3**

Also transfer these values to the Risk Assessment Matrices for soils, surface water and groundwater.

Soil

Texture

Low vulnerability: soil texture is loam to medium clay.

Medium vulnerability: soil texture is duplex with a light topsoil and a heavy subsoil or is heavy clay.

High vulnerability: soil texture is sand or unknown.

The soil texture of the effluent irrigation area is light clay, thus the risk weighting is low (**1**).

Depth

Low vulnerability: Depth of soil is > 1 m.

Medium vulnerability: Depth of soil is 0.5 – 1m.

High vulnerability: Depth of soil is < 0.5 m or unknown.

The soil depth of the effluent irrigation area is several meters, thus the risk weighting is low (**1**).

Slope

Low vulnerability: Slope is < 5% or slope is 5-10% but continuous vegetative cover is constantly maintained over the area or slope is 5-10% but a system of well-designed contour banks is in place to slow the movement of water from the site.

Medium vulnerability: Slope is 5 – 10% or slope is >10% but continuous vegetative cover is constantly maintained over the area or slope is >10% but a system of well-designed contour banks is in place to slow the movement of water from the site.

High vulnerability: Slope is > 10% or unknown.

The effluent irrigation area is located on floodplains that are almost flat, thus the risk weighting is low (**1**).

Soil Dispersion

Low vulnerability: Soil does not disperse on wetting and has a low exchangeable sodium percentage (less than 6%).

Medium vulnerability: Soil disperses on wetting and / or has an exchangeable sodium percentage of 6-15%.

High vulnerability: Soil disperses on wetting and / or has an exchangeable sodium percentage exceeding 15% or the dispersive behaviour and exchangeable sodium percentage of the soil are unknown.

The exchangeable sodium percentage of the site can be calculated as 3% in the 0-10 cm layer, 8.5% in the 20-30 cm layer and 11.1% in the 50-60 cm layer. Thus the risk weighting is medium (**2**).

Salinity

Low vulnerability: Soil is in the very low to low salinity class (EC_{se} is less than 1.9 dS/m)

Medium vulnerability: Soil is in the medium salinity class (EC_{se} is 1.9-4.5 dS/m)

High vulnerability: Soil is in the high to extreme salinity class (EC_{se} is over 4.5 dS/m) or soil salinity class is unknown.

The soil has a low to very low salinity class to a depth of 60 cm, thus the risk weighting is low (**1**).

Nutrient Status

Nitrogen

Low vulnerability: Either soil solution nitrate-N levels at the base of the active root zone are <10 mg/L or are less than measured baseline data.

High vulnerability: Either soil solution nitrate-N levels at the base of the active root zone are >10 mg/L or are greater than measured baseline data.

These can be converted to soil nitrate-nitrogen concentrations for different soil types as per Table 31 (Section 8.1.7) of the Resource Manual.

Nitrate-N concentrations measured to a depth of 60 cm exceed the recommended maximum 2.5 mg/kg of nitrate-N at the base of the root zone for a light clay, thus the risk is high (**3**).

Phosphorus

Vulnerability ratings for phosphorus are based on three methods.

Method 1 involves a check as to whether the Colwell Extractable phosphorus levels exceed certain limits. These limits are based on measured Colwell extractable phosphorus for numerous soils (categorised by clay content and pH). The upper limits (high rating) are one standard deviation above the mean of numerous Colwell extractable phosphorus levels (Redding, 2002). However, these limits may not be appropriate for some soil types, such as black vertosols, which may have high levels of Colwell phosphorus in their 'virgin' state.

Method 1 (Most Soils)

Low vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	< 15
< 30%	> 7	< 30
> 30%	< 7	< 40
> 30%	> 7	< 45

Medium vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	15 – 30
< 30%	> 7	30 – 60
> 30%	< 7	40 – 75
> 30%	> 7	45 – 85

High vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	> 30
< 30%	> 7	> 60
> 30%	< 7	> 75
> 30%	> 7	> 85

Colwell extractable P levels have been measured at 40 mg/kg in the top 10 cm. The topsoil has a pH of 5.3 and we can assume the clay content is greater than 30%, thus the risk weighting is medium (**2**).

Surface Water

Surface water includes water in dams, reservoirs, rivers, creeks and all other waterways where rainfall is likely to collect. Ideally, reuse areas should be well separated from surface water bodies, particularly those used for sensitive purposes e.g. town water supplies. However, distance is not the only criterion determining the potential for contamination from reuse areas. Design and management factors, particularly the amount and type of vegetative cover, may significantly reduce any potential contamination of surface waters.

Water Quality Protection

Low vulnerability: Reuse area is at least 200 m from a surface water body and effluent irrigations do not cause runoff <u>or</u> is at least 150 m from a surface water body but includes a vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff <u>or</u> is at least 100 m from a surface water body but includes a well-maintained vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff or there is a terminal pond sized to catch the first 12 mm of rainfall runoff plus irrigation water runoff.

Medium vulnerability: Reuse area is between 100 m and 200 m from a surface water body and effluent irrigations do not cause runoff or is at least 75 m from a surface water body but includes a vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff or is at least 50 m from a surface water body but includes a well-maintained vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff.

High vulnerability: Reuse area has no vegetative buffer and is less than 100 m from a surface water body or reuse area has a vegetative buffer but is within 50 m of a surface water body or and effluent irrigations create runoff that is not captured in a terminal pond.

The effluent irrigation area is not within 200 m of the closest surface water body and effluent irrigations do not cause runoff. The surface water protection measures in place match the risk weighting of low (**1**).

Flood potential

Low vulnerability: reuse area is above the 1 in 10 year flood line.

Medium vulnerability: reuse area is above the 1 in 5 year flood line but below the 1 in 10 year flood line.

High vulnerability: reuse area is below the 1 in 5 year flood line or flooding frequency of reuse area is unknown.

The flood frequency is less than 1 in 10 years, thus the risk weighting is low (**1**).

Groundwater

Ideally, reuse areas should be located on areas with deep groundwater or on those well protected by a clay blanket or confined aquifer. The risk to groundwater from effluent reuse depends upon the protection afforded by soil type (e.g. a deep clay blanket may afford good protection, a sandy loam soil provides relatively poor protection) and the geology and type of aquifer (e.g a confined aquifer versus an alluvial aquifer).

The consequences of nutrient or salt leaching to groundwater depend on the quality of the groundwater (e.g. potable water V brackish water).

Depth to groundwater

Low vulnerability: Groundwater is at least 20 m below the surface, or there is no significant groundwater.

Medium vulnerability: Groundwater is 10-20 m below the surface.

High vulnerability: Groundwater is less than 10 m below the surface <u>or</u> depth to groundwater is unknown.

There is no significant groundwater below the property, thus the risk weighting is low (**1**).

Soil type

Low vulnerability: There is at least 0.5 m of clay above the aquifer or the aquifer is confined, <u>or</u> there is no significant groundwater.

Medium vulnerability: There is at least a metre of loam to clay soil above the aquifer.

High vulnerability: The soil above the aquifer is sand, sandy-loam or gravel <u>or</u> there is less than a metre of soil above the aquifer <u>or</u> the soil type above the aquifer is unknown.

There is no significant groundwater, thus the risk weighting is low (**1**).

Water quality

Low vulnerability: The groundwater resources in the area are of a quality having no productive use e.g. EC exceeds 8 dS/m, <u>or</u> there is no significant groundwater.

Medium vulnerability: Groundwater resources are suitable for stock drinking water and irrigation e.g. EC of up to 8 dS/m & containing less than 100 mg NO_3N/L

High vulnerability: Groundwater resources are suitable for human consumption. (EC of up to 1.6 dS/m and containing less than 10 mg NO_3N/L) <u>or</u> the quality of groundwater resources is unknown.

There is no significant groundwater beneath the site. Hence, the risk weighting is low (**1**).

Table 68, Table 69 and Table 53 are templates for recording the site vulnerability risk weightings for soil, surface water and groundwater. To complete the tables, a vulnerability weighting of 1, 2 or 3 applies to each sub-category of soil, surface water and groundwater. A low vulnerability attracts a vulnerability weighting of “1”, medium vulnerability attracts a vulnerability weighting of “2” and high vulnerability attracts a vulnerability weighting of “3”. These numbers are transferred to Table 54, Table 72 and Table 73.

Risk Assessment Tables

TABLE 68 - VULNERABILITY WEIGHTINGS - SOIL

Resource	Texture (weighting low = 1, med. = 2, high = 3)	Depth (weighting low = 1, med. = 2, high = 3)	Slope (weighting low = 1, med. = 2, high = 3)	Soil Dispersion (weighting low = 1, med. = 2, high = 3)	Salinity (weighting low = 1, med. = 2, high = 3)	Nitrogen (weighting low = 1, med. = 2, high = 3)	Phosphorus (weighting low = 1, med. = 2, high = 3)
Site Vulnerability Weighting	**1**	**1**	**1**	**2**	**1**	**3**	**2**

TABLE 69 – VULNERABILITY WEIGHTINGS – SURFACE WATER

Resource	Water Quality Protection Weighting (low = 1, medium = 2, high = 3)	Flood Potential Weighting (low = 1, medium = 2, high = 3)
Site Vulnerability Weighting	**1**	**1**

TABLE 70 – VULNERABILITY WEIGHTINGS - GROUNDWATER

Resource	Depth to Groundwater Weighting (low = 1, medium = 2, high = 3)	Soil Type Weighting (low = 1, medium = 2, high = 3)	Water Quality Weighting (low = 1, medium = 2, high = 3)
Site Vulnerability Weighting	**1**	**1**	**1**

Transfer these values to the Risk Assessment Matrix Tables.

TABLE 71 - RISK ASSESSMENT MATRIX - SOIL

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Texture	Depth	Slope	Soil Dispersion	Salinity	Nitrogen	Phosphorus
		1	**1**	**1**	**2**	**1**	**3**	**2**
Nutrients in manure and effluent	**1**	**1**	**1**	**1**	**2**	**1**	**3**	**2**
Size of land area	**2**	**2**	**2**	**2**	**4**	**2**	**6**	**4**
Application method	**3**	**3**	**3**	**3**	**6**	**3**	**9**	**6**

TABLE 72 - RISK ASSESSMENT MATRIX – SURFACE WATER

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Site Vulnerability Weighting	
		Water Quality Protection	Flood Potential
		1	**1**
Nutrients in manure and effluent	**1**	**1**	**1**
Size of land area	**2**	**2**	**2**
Application method	**3**	**3**	**3**

TABLE 73 - RISK ASSESSMENT MATRIX - GROUNDWATER

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Site Vulnerability Weighting		
		Depth	Soil Type	Water Use
		1	**1**	**1**
Nutrients in manure and effluent	**1**	**1**	**1**	**1**
Size of land area	**2**	**2**	**2**	**2**
Application methods	**3**	**3**	**3**	**3**

Monitoring Based on Risk

Ratings of 1 and 2 require minimal monitoring. Ratings of 3, 4 & 6 attract moderate levels of monitoring. A rating of 9 requires intensive monitoring.

At this piggery, five ratings of 3, two ratings of four, two ratings of 6 and one rating of 9 have been determined for soils. The existing nutrient status and ESP of the soil, along with the reuse method, has driven these results. A rating of 3 has been achieved for surface water due mainly to the risk associated with the application method. For groundwater, no ratings above 3 were obtained. Because a rating of 9 was calculated for soil, intensive monitoring would be warranted for the soil.

Soils

Where the risk of soil related impacts is low (rating of 1-3) and at least 3 years of annual monitoring shows that the system is sustainable, it is suggested that soils from reuse areas should be monitored at least every three years. Those in a low risk category will not need to monitor effluent quality *unless* they are already undertaking this monitoring (which is the reason for being in this category).

Where there is a medium risk of soil impacts (rating of 4 or 6) and at least 3 years of monitoring data show that the system is sustainable, it is suggested that soils from reuse areas should be sampled and analysed at least every two years. Effluent and solids quality (if reused on-site) should also be analysed annually.

Where there is a high risk of soil impacts (rating of 9), annual soil monitoring is imperative. Effluent and solids quality (if reused on-site) should also be analysed annually.

Surface Water

Surface water quality monitoring is not suggested as a relevant measure of sustainability for piggeries and cattle feedlots, as they are not direct discharge industries (e.g. sewage treatment plants) and generally rely on land application for the reuse of by-products. To be able to achieve any meaningful results from a monitoring perspective, surface water monitoring would require sophisticated equipment and trained operators.

Piggeries and cattle feedlots are required to comply with relevant codes of practice for their design and management, such as appropriate buffers, vegetative filter strips or terminal ponds. If an enterprise attracts a high rating, remedial action in the form of improved design and/or management of the reuse are would be required.

Groundwater

Groundwater quality monitoring would be warranted for anyone attracting a high rating (9). Ideally this would include sampling and analysis from bores upslope and downslope of reuse areas. Electrical conductivity and nitrate-nitrogen should be determined. On very sandy soils, total P should also be measured. If a moderate risk weighting is attracted for groundwater, monitoring would not necessarily be required, provided nutrient and salt risk weightings for the soil are low.

Soil monitoring

The risk weightings for soils comprise a mixture of low (1 or 2), medium (3, 4 or 6) and high (9). The medium to high results are due to the interaction between size of land area (2) or application method (3) and soil dispersion (2) or nutrient status (3 for nitrogen and 2 for phosphorus). The risk could be significantly reduced by verifying the size of the land area, by changing to a travelling irrigator and by spelling the area for a time, expanding it or cropping it more intensively to reduce the nutrient content (particularly nitrogen).

Surface water monitoring

The surface water risk assessment produced low levels of risk. For this particular site, no further remedial action in the form of improved design/management is warranted.

Groundwater monitoring

The groundwater risk assessment produced low levels of risk, due to the absence of groundwater. No monitoring of groundwater should be required.

Appendix H. EXAMPLE RISK ASSESSMENT - FEEDLOT

Description of Development

Capacity: 5000 SCU

Property size: 725 ha

Solid manure: All sold off-site to neighboring landholders and compost plant

Feedlot Site Description: The feedlot is constructed on a ridge of approximately 3% natural slope. The soils on the ridge have a gravely loam topsoil with a heavy clay underneath.

Irrigation area description: The effluent irrigation area is 43 ha located on the alluvial flats. The soils comprise deep, uniform, medium cracking clays formed on alluvial materials. A granular self-mulching layer forms at the surface when dry and this is underlain by fine to medium blocky structure. The soil pH ranges from medium acid to neutral at the surface, gradually decreasing in acidity to be strongly alkaline at depth. The soils have moderate to high fertility and high available moisture storage capacities. Soil sampling revealed that the soil depth is at least 60 cm.

TABLE 74 - ANALYSIS OF SOIL IN THE EFFLUENT IRRIGATION AREA.

Parameter	Unit	Depth (cm)	Results
PH		0 – 10	5.8
		20 – 30	7.6
		50 - 60	7.8
Total-N	N mg/kg	0 - 10	2600
Nitrogen (Nitrate)	N mg/kg	0 – 10	5
		20 – 30	3
		50 - 60	3
Phosphorus (Colwell)	P mg/kg	0 - 10	13
Conductivity	dS/m	0 – 10	0.05
		20 – 30	0.13
		50 - 60	0.18
ESP	%	0 - 10	0.4

P storage: Based on a 20 year reuse life, the site can safely store 50 kg/ha/yr of P. This was calculated from the P sorption isotherm, using a P solution concentration of 0.5 mg P/L.

Surface water: A creek that runs only intermittently drains through the centre of the property. Effluent irrigation is not undertaken within 50 m of the creek. A vegetative filter strip of well-maintained grass and remnant vegetation lies between the edge of the effluent irrigation area and the top of the creek bank.

Groundwater: A number of bores have been drilled on the property, with the availability of groundwater supplies being unreliable. At one bore close to the creek on the alluvium, groundwater was obtained at a depth of 17m in sand and gravel. The effluent irrigation area is located on grey vertisol appears on the margins of the flood plain. The high clay content of the soil would provide protection against nutrient leaching. However, the sand and gravel that is sometimes present beneath this soil type would drain readily. Thus, careful irrigation and crop management is essential to ensure there is no contamination of any alluvial water.

Effluent storage: Runoff from the feedlot flows through a sedimentation basin to remove the settled solids before it flows into an effluent treatment pond designed to avoid overtopping more than 1 in 10 years on average.

Effluent irrigation: The MEDLI model has been used to predict the quantity and quality of effluent for irrigation from the feedlot. MEDLI predicts that approximately 37 ML of effluent will need to be irrigated annually onto the 43 ha effluent irrigation area. The effluent applied in irrigation will contain of 16.4 t of N, 2.4 t of P and 53.2 t of TDS. The predicted concentrations of effluent from the pond are 476 mg/L of N, 116 mg/L of P and 0.71 dS/m EC. An additional 32.5 ML/yr of fresh shandying water will be mixed with the effluent for irrigation. Terminal ponds have been constructed to catch the first 25 mm of runoff from the irrigation area. Effluent will be applied with a low pressure traveling irrigator.

Crop production: Growing a forage sorghum crop, MEDLI predicts the dry matter yield will be 16.9 t/ha/yr, with N removal of 221 kg/ha/yr and P removal of 96.2 kg/ha/yr. The SALF model was used to predict the effect of effluent irrigation on root zone salinity, with an average root zone ECse of 0.87 dS/m and no effect on crop yield.

Risk Assessment

Risk Assessment: Nutrients in Manure and Effluent

Low Risk The quantity of effluent and solids reused is measured and the quality of effluent and solids reused is regularly measured (at least annually, more frequently if required to ensure sound management of nutrients). OR

You have developed a mass balance of nutrient production from your piggery or cattle feedlot using accepted design tools, such as PigBal, BeefBal or MEDLI using conservative figures. (There can be a great variation in nutrient predictions from mass balance models).

High Risk You have never measured, but only estimated the mass of nutrients applied using “text-book” values, such as those provided in Section 6.3.12 (piggeries) and 6.6.9 (feedlots) of the Resource Manual.

A risk weighting of 1 or 3 applies to the Nutrients and Manure criterion. A low risk attracts a risk weighting of “1” and high risk attracts a risk weighting of “3”.

Transfer these values to the Risk Assessment Matrices for soils, surface water and groundwater.

As a mass balance of nutrient effluent application has been conducted, the risk assessment of nutrients in manure and effluent is low, so a number of **1** can be applied.

Risk Assessing the Site

Transfer data from this assessment into the Design and Management Reuse Area Risk Assessment Summary table.

Size of Land Area

Knowledge of Size of Land Area

Low Risk	From farm or paddock maps, you accurately know the area (ha) of each effluent or manure reuse paddock under each management regime (e.g. soil properties, land use).
Medium Risk	You know the approximate area (ha) of each effluent or manure reuse paddock under each management regime.
High Risk	You do not know the area of the effluent or manure reuse paddocks.

The area of the effluent irrigation area is approximately known from aerial photography, so the risk weighting is medium (2).

Knowledge of Yields of Crops or Pastures Grown on Reuse Areas

Low Risk	For your property and soil type, you know typical yields for the pastures or crops grown on reuse areas.
Medium Risk	You know typical district yields for the pastures or crops grown on reuse areas.
High Risk	You do not know typical yields for the pastures or crops grown on reuse areas.

MEDLI contains an in-built crop model and the modelling estimates dry matter yields that are typical of the previous farm production, thus the risk weighting is low (**1**).

Knowledge of Nutrients Applied to Reuse Areas

Low Risk	You have calculated the nitrogen (kg/ha/yr) and phosphorus (kg/ha/yr) loading rates to reuse areas from estimated nutrient production.
High Risk	You have not calculated the nitrogen (kg/ha/yr) and phosphorus (kg/ha/yr) loading rates to reuse areas.

The nitrogen and phosphorus to be applied has been calculated, thus the risk weighting is low (**1**).

Nitrogen Mass Balance for Reuse Areas

Low Risk	You have calculated that the net mass of nitrogen (kg/ha/yr) applied as effluent and / or solid by-products is exceeded by the mass of nitrogen (kg/ha/yr) that plant harvest should remove.
Medium Risk	You have calculated that the net mass of nitrogen applied (kg/ha/yr) as effluent or solid by-products is equal to the mass of nitrogen (kg/ha/yr) that plant harvest should remove.
High Risk	The net mass of nitrogen applied to reuse areas (kg/ha/yr) exceeds the mass removed <u>or</u> you do not know the net mass of nitrogen applied to the reuse area.

The MEDLI modelling shows that the amount of N removed by the crop should be 221 kg/ha/yr and the amount applied is only 189.3 kg/ha/yr, thus the risk weighting is low (**1**).

Phosphorus Mass Balance for Reuse Areas

Low Risk	You have calculated that the mass of phosphorus (kg/ha/yr) applied as effluent and / or solid by-products is exceeded by the mass of phosphorus (kg/ha/yr) that plant harvest should remove plus phosphorus storage calculated from a site-specific phosphorus sorption test.
Medium Risk	You have calculated that the net mass of phosphorus applied (kg/ha/yr) as effluent or solid by-products is equal to the mass of phosphorus (kg/ha/yr) that plant harvest should remove plus phosphorus storage calculated from a site-specific phosphorus sorption test or from generic phosphorus sorption data for similar soil types.
High Risk	The net mass of phosphorus applied to reuse areas (kg/ha/yr) exceeds the mass removed plus storage calculated from a site-specific phosphorus sorption test or from generic phosphorus sorption data for similar soil types <u>or</u> you do not know the mass of phosphorus applied to the reuse area.

The MEDLI modelling shows that the amount of P removed by the crop should be 96.2 kg/ha/yr and the safe P storage capacity is 50 kg/ha/yr from site-specific P sorption data. The amount of P in effluent is only 109.8 kg/ha/yr, thus the risk weighting is low (**1**).

Using Appropriate Effluent & Solid By-Product Application Methods

If you reuse effluent on-site, select the appropriate risk category for “Effluent Irrigation” based on the information presented below. If you reuse solid by-products on-site, select the appropriate risk category for “Solids Spreading” from the information presented below. If you reuse effluent and solids on the same area, select the risk weighting that is highest from either the “Effluent Irrigation” or “Solids Spreading” sections below (e.g. if you have a rating of *low* for effluent irrigation and a rating of *medium* for solids spreading, the overall risk weighting you choose for the area is *medium*).

The results then need to be transferred into the “Using Appropriate Effluent & Solid By-Product Application Methods” row of Table 30 and converted into a risk weighting. A separate copy of Table 30 needs to be developed for separate reuse areas (e.g. effluent areas V solid areas) or reuse areas posing different risks (e.g. one effluent reuse area might be low risk, another high risk).

Effluent Irrigation

Low Risk — You use a low-pressure, travelling spray or drip irrigation system or a low-pressure solid set spray or drip irrigation system or a well designed and maintained flood irrigation system that is not on sandy to sandy loam soil. The system also applies effluent evenly and at target rates.

High Risk — You use a hand-shift sprinkler or hose or a poorly designed or managed flood irrigation system (e.g. land has not been levelled or effluent is unshandied or surface soil is sandy to sandy loam).

Effluent irrigation will be applied evenly with a low-pressure travelling irrigator, thus the risk weighting is low (**1**).

Solids Spreading

Low Risk — The spreading method used disperses solids evenly and at target rates.

Medium Risk — The spreading method used disperses solids fairly evenly and within 20% of target rates.

High Risk — The spreading method used disperses solids unevenly or at uncontrolled rates (not within 20% of target rates).

Table 67 is a template for summarising the design and management risk weightings for each design and management criterion. To complete the table, insert a risk weighting of 1, 2 or 3 against each criterion. A low risk attracts a risk weighting of "1", medium risk attracts a risk weighting of "2" and high risk attracts a risk weighting of "3". These numbers are transferred to Table 54, Table 72 and Table 73.

All solid manure will be sold off-farm.

TABLE 75 - DESIGN AND MANAGEMENT REUSE AREA RISK ASSESSMENT SUMMARY

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)
Size of land area	**2**
Application methods	**1**

Also transfer these values to the Risk Assessment Matrices for soils, surface water and groundwater.

Soil

Texture

Low vulnerability: soil texture is loam to medium clay.

Medium vulnerability: soil texture is duplex with a light topsoil and a heavy subsoil or is heavy clay.

High vulnerability: soil texture is sand or unknown.

The soil texture of the effluent irrigation area is a medium clay, thus the risk weighting is low (**1**).

Depth

Low vulnerability: Depth of soil is > 1 m.

Medium vulnerability: Depth of soil is 0.5 – 1m.

High vulnerability: Depth of soil is < 0.5 m or unknown.

The soil depth of the effluent irrigation area is at least 60 cm, but it is not known if is greater than a metre, thus the risk weighting is medium (**2**).

Slope

Low vulnerability: Slope is < 5% or slope is 5-10% but continuous vegetative cover is constantly maintained over the area or slope is 5-10% but a system of well-designed contour banks is in place to slow the movement of water from the site.

Medium vulnerability: Slope is 5 – 10% or slope is >10% but continuous vegetative cover is constantly maintained over the area or slope is >10% but a system of well-designed contour banks is in place to slow the movement of water from the site.

High vulnerability: Slope is > 10% or unknown.

The effluent irrigation area is located on the alluvial flats and is safe to assume the slope is less than 5%, thus the risk weighting is low (**1**).

Soil Dispersion

Low vulnerability: Soil does not disperse on wetting and has a low exchangeable sodium percentage (less than 6%).

Medium vulnerability: Soil disperses on wetting and / or has an exchangeable sodium percentage of 6-15%.

High vulnerability: Soil disperses on wetting and / or has an exchangeable sodium percentage exceeding 15% or the dispersive behaviour and exchangeable sodium percentage of the soil are unknown.

The exchangeable sodium percentage of the site is 0.4%, thus the risk weighting is low (**1**).

Salinity

Low vulnerability: Soil is in the very low to low salinity class (EC_{se} is less than 1.9 dS/m)

Medium vulnerability: Soil is in the medium salinity class (EC_{se} is 1.9-4.5 dS/m)

High vulnerability: Soil is in the high to extreme salinity class (EC_{se} is over 4.5 dS/m) or soil salinity class is unknown.

The soil has a low to very low salinity class to depths of 60 cm, thus the risk weight is low (**1**).

Nutrient Status

Nitrogen

Low vulnerability: Either soil solution nitrate-N levels at the base of the active root zone are <10 mg/L or are less than measured baseline data.

High vulnerability: Either soil solution nitrate-N levels at the base of the active root zone are >10 mg/L or are greater than measured baseline data.

These can be converted to soil nitrate-nitrogen concentrations for different soil types as per Table 31 (Section 8.1.7) of the Resource Manual.

Nitrate-N concentrations measured to a depth of 60 cm are less than the recommended maximum 3.5 mg/kg of nitrate-N at the base of the root zone for a medium clay, thus the risk is low (**1**).

Phosphorus

Vulnerability ratings for phosphorus are based on three methods.

Method 1 involves a check as to whether the Colwell Extractable phosphorus levels exceed certain limits. These limits are based on measured Colwell extractable phosphorus for numerous soils (categorised by clay content and pH). The upper limits (high rating) are one standard deviation above the mean of numerous Colwell extractable phosphorus levels (Redding, 2002). However, these limits may not be appropriate for some soil types, such as black vertosols, which may have high levels of Colwell phosphorus in their 'virgin' state.

Method 1 (Most Soils)

Low vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	< 15
< 30%	> 7	< 30
> 30%	< 7	< 40
> 30%	> 7	< 45

Medium vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	15 – 30
< 30%	> 7	30 – 60
> 30%	< 7	40 – 75
> 30%	> 7	45 – 85

High vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	> 30
< 30%	> 7	> 60
> 30%	< 7	> 75
> 30%	> 7	> 85

Colwell extractable P levels have been measured at 13 mg/kg in the top 10 cm. The topsoil has a pH of 5.8 and we can assume the clay content is greater than 30%, thus the risk weighting is low (**1**).

Surface Water

Surface water includes water in dams, reservoirs, rivers, creeks and all other waterways where rainfall is likely to collect. Ideally, reuse areas should be well separated from surface water bodies, particularly those used for sensitive purposes e.g. town water supplies. However, distance is not the only criterion determining the potential for contamination from reuse areas. Design and management factors, particularly the amount and type of vegetative cover, may significantly reduce any potential contamination of surface waters.

Water Quality Protection

Low vulnerability: Reuse area is at least 200 m from a surface water body and effluent irrigations do not cause runoff <u>or</u> is at least 150 m from a surface water body but includes a vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff <u>or</u> is at least 100 m from a surface water body but includes a well-maintained vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff <u>or</u> there is a terminal pond sized to catch the first 12 mm of rainfall runoff plus irrigation water runoff.

Medium vulnerability: Reuse area is between 100 m and 200 m from a surface water body and effluent irrigations do not cause runoff or is at least 75 m from a surface water body but includes a vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff or is at least 50 m from a surface water body but includes a well-maintained vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff.

High vulnerability: Reuse area has no vegetative buffer and is less than 100 m from a surface water body or reuse area has a vegetative buffer but is within 50 m of a surface water body or and effluent irrigations create runoff that is not captured in a terminal pond.

The effluent irrigation area has a 50m wide, well-maintained buffer and terminal ponds are in place to catch the first 25mm of runoff. The surface water protection measures in place match the risk weighting of low (**1**) – terminal ponds are in place.

Flood potential

Low vulnerability: reuse area is above the 1 in 10 year flood line.

Medium vulnerability: reuse area is above the 1 in 5 year flood line but below the 1 in 10 year flood line.

High vulnerability: reuse area is below the 1 in 5 year flood line or flooding frequency of reuse area is unknown.

The flood frequency is unknown, but the area regularly floods, thus the risk weighting is high (**3**).

Groundwater

Ideally, reuse areas should be located on areas with deep groundwater or on those well protected by a clay blanket or confined aquifer. The risk to groundwater from effluent reuse depends upon the protection afforded by soil type (e.g. a deep clay blanket may afford good protection, a sandy loam soil provides relatively poor protection) and the geology and type of aquifer (e.g a confined aquifer versus an alluvial aquifer).

The consequences of nutrient or salt leaching to groundwater depend on the quality of the groundwater (e.g. potable water V brackish water).

Depth to groundwater

Low vulnerability: Groundwater is at least 20 m below the surface, or there is no significant groundwater.

Medium vulnerability: Groundwater is 10-20 m below the surface.

High vulnerability: Groundwater is less than 10 m below the surface or depth to groundwater is unknown.

Depth to groundwater is known to be about 17m, thus the risk weighting is medium (**2**).

Soil type

Low vulnerability: There is at least 0.5 m of clay above the aquifer or the aquifer is confined, or there is no significant groundwater.

Medium vulnerability: There is at least a metre of loam to clay soil above the aquifer.

High vulnerability: The soil above the aquifer is sand, sandy-loam or gravel or there is less than a metre of soil above the aquifer or the soil type above the aquifer is unknown.

Bore logs suggest there is at least 1m of clay above the aquifer, thus the risk weighting is low (**1**).

Water quality

Low vulnerability: The groundwater resources in the area are of a quality having no productive use e.g. EC exceeds 8 dS/m, or there is no significant groundwater.

Medium vulnerability: Groundwater resources are suitable for stock drinking water and irrigation e.g. EC of up to 8 dS/m & containing less than 100 mg NO_3N/L

High vulnerability: Groundwater resources are suitable for human consumption. (EC of up to 1.6 dS/m and containing less than 10 mg NO_3N/L) or the quality of groundwater resources is unknown.

Tests of bores suggest that the bore water in the area is of high quality and is suitable for human consumption, thus the risk weighting is high (**3**).

Table 68, Table 69 and Table 53 are templates for recording the site vulnerability risk weightings for soil, surface water and groundwater. To complete the tables, a vulnerability weighting of 1, 2 or 3 applies to each sub-category of soil, surface water and groundwater. A low vulnerability attracts a vulnerability weighting of “1”, medium vulnerability attracts a vulnerability weighting of “2” and high vulnerability attracts a vulnerability weighting of “3”. These numbers are transferred to Table 54, Table 72 and Table 73.

Risk Assessment Tables

TABLE 76 - VULNERABILITY WEIGHTINGS - SOIL

Resource	Texture (weighting low = 1, med. = 2, high = 3)	Depth (weighting low = 1, med. = 2, high = 3)	Slope (weighting low = 1, med. = 2, high = 3)	Soil Dispersion (weighting low = 1, med. = 2, high = 3)	Salinity (weighting low = 1, med. = 2, high = 3)	Nitrogen (weighting low = 1, med. = 2, high = 3)	Phosphorus (weighting low = 1, med. = 2, high = 3)
Site Vulnerability Weighting	**1**	**2**	**1**	**1**	**1**	**1**	**1**

TABLE 77 – VULNERABILITY WEIGHTINGS – SURFACE WATER

Resource	Water Quality Protection Weighting (low = 1, medium = 2, high = 3)	Flood Potential Weighting (low = 1, medium = 2, high = 3)
Site Vulnerability Weighting	**1**	**3**

TABLE 78 – VULNERABILITY WEIGHTINGS - GROUNDWATER

Resource	Depth to Groundwater Weighting (low = 1, medium = 2, high = 3)	Soil Type Weighting (low = 1, medium = 2, high = 3)	Water Quality Weighting (low = 1, medium = 2, high = 3)
Site Vulnerability Weighting	**2**	**1**	**3**

Transfer these values to the Risk Assessment Matrix Tables.

TABLE 79 - RISK ASSESSMENT MATRIX - SOIL

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Texture	Depth	Slope	Soil Dispersion	Salinity	Nitrogen	Phosphorus
		1	**2**	**1**	**1**	**1**	**1**	**1**
Nutrients in manure and effluent	**1**	**1**	**2**	**1**	**1**	**1**	**1**	**1**
Size of land area	**2**	**2**	**4**	**2**	**2**	**2**	**2**	**2**
Application method	**1**	**1**	**2**	**1**	**1**	**1**	**1**	**1**

TABLE 80 - RISK ASSESSMENT MATRIX – SURFACE WATER

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Site Vulnerability Weighting	
		Water Quality Protection	Flood Potential
		1	**3**
Nutrients in manure and effluent	**1**	**1**	**3**
Size of land area	**2**	**2**	**6**
Application method	**1**	**1**	**3**

TABLE 81 - RISK ASSESSMENT MATRIX - GROUNDWATER

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Site Vulnerability Weighting		
		Depth	Soil Type	Water Use
		2	**1**	**3**
Nutrients in manure and effluent	**1**	**2**	**1**	**3**
Size of land area	**2**	**4**	**2**	**6**
Application methods	**1**	**2**	**1**	**3**

Monitoring based on Risk

Ratings of 1 and 2 require minimal monitoring. Ratings of 3, 4 & 6 attract moderate levels of monitoring. A rating of 9 requires intensive monitoring. It is important to realise that if a rating of 4 is calculated for groundwater and a rating of 9 is calculated for soil, moderate monitoring would be warranted for the groundwater and intensive monitoring would be warranted for the soil.

Soils

Where the risk of soil related impacts is low (rating of 1-3) and at least 3 years of annual monitoring shows that the system is sustainable, it is suggested that soils from reuse areas should be monitored at least every three years. Those in a low risk category will not need to monitor effluent quality *unless* they are already undertaking this monitoring (which is the reason for being in this category).

Where there is a medium risk of soil impacts (rating of 4 or 6) and at least 3 years of monitoring data show that the system is sustainable, it is suggested that soils from reuse areas should be sampled and analysed at least every two years. Effluent and solids quality (if reused on-site) should also be analysed annually.

Where there is a high risk of soil impacts (rating of 9), annual soil monitoring is imperative. Effluent and solids quality (if reused on-site) should also be analysed annually.

Surface Water

Surface water quality monitoring is not suggested as a relevant measure of sustainability for piggeries and cattle feedlots, as they are not direct discharge industries (e.g. sewage treatment plants) and generally rely on land application for the reuse of by-products. To be able to achieve any meaningful results from a monitoring perspective, surface water monitoring would require sophisticated equipment and trained operators.

Piggeries and cattle feedlots are required to comply with relevant codes of practice for their design and management, such as appropriate buffers, vegetative filter strips or terminal ponds. If an enterprise attracts a high rating, remedial action in the form of improved design and/or management of the reuse are would be required.

Groundwater

Groundwater quality monitoring would be warranted for anyone attracting a high rating (9). Ideally this would include sampling and analysis from bores upslope and downslope of reuse areas. Electrical conductivity and nitrate-nitrogen should be determined. On very sandy soils, total P should also be measured. If a moderate risk weighting is attracted for groundwater, monitoring would not necessarily be required, provided nutrient and salt risk weightings for the soil are low.

Soil monitoring

All of the risk weightings for soils are low (1 or 2), except for the size of land area/soil depth (4). However, both of these weightings could be reduced to low (1) by accurately measuring the size of the effluent irrigation area and verify the total depth of soil. Thus, for this site if 3 years of monitoring revealed no change in the triggers for sustainability, the frequency of monitoring could be reduced.

Surface water monitoring

The surface water risk assessment produced moderate levels of risk, due to the flooding potential of the site. This however, is off-set by the surface water protection measures that are in place (vegetative buffer areas and terminal ponds).

Groundwater monitoring

The groundwater risk assessment produced moderate levels of risk, due mainly to the high quality groundwater of the area and uncertainty over the size of the reuse area. These risks could be reduced by accurately knowing the size of the application area. However, because the site generally has low risks associated with the soils, groundwater monitoring should not be a prerequisite while the soil risk weightings remain low. Monitoring of groundwater may be required if the risk weightings of the soil increased to medium or high.

Appendix I. EXAMPLE RISK ASSESSMENT – FEEDLOT 2

Description of Development

Capacity: 5000 SCU

Property size: 725 ha

Solid manure: All sold off-site to neighboring landholders and compost plant

Feedlot Site Description: The feedlot is constructed on a ridge of approximately 3% natural slope. The soils on the ridge are shallow gravelly sandy loams.

Irrigation area description: The effluent irrigation area is approximately 100 ha of deep sandy loam flats. The soil depth is at least 1 m. The nutrient status of the soils of the reuse area is shown in Table 82.

TABLE 82 - ANALYSIS OF SOIL IN THE EFFLUENT IRRIGATION AREA.

Parameter	Unit	Depth (cm)	Results
PH		0 – 10	5.8
		20 – 30	5.6
		50 - 60	4.8
Total-N	N mg/kg	0 - 10	1300
Nitrogen (Nitrate)	N mg/kg	0 – 10	2
		20 – 30	3
		50 - 60	4
Phosphorus (Colwell)	P mg/kg	0 - 10	17
Conductivity	dS/m	0 – 10	0.05
		20 – 30	0.08
		50 - 60	0.08
ESP	%	0 - 10	10

P storage: The phosphorus storage capacity of the soil is unknown.

Surface water: A creek runs through the centre of the property. Effluent irrigation is not undertaken within 100 m of the creek. The area has a flooding frequency of about one in ten years.

Groundwater: There are several bores on the property. Groundwater is present at a depth of approximately 40-50 m. Drilling logs for the bores are not available. However, it is known that there is deep clay under the sandy loam soil.

Effluent storage: Runoff from the feedlot flows through a sedimentation basin to remove the settled solids before it flows into an effluent treatment pond designed to avoid overtopping more than 1 in 10 years on average.

Effluent irrigation: Effluent will be applied with a low pressure traveling irrigator.

Crop production: Forage sorghum crops grown on the property typically yield 12 t DM/ha/yr, MEDLI predicts the dry matter yield will be 14.5 tha/yr, with N removal of 157 kg/ha/yr and P removal of 69 kg/ha/yr.

Risk Assessment

Risk Assessment: Nutrients in Manure and Effluent

Low Risk The quantity of effluent and solids reused is measured and the quality of effluent and solids reused is regularly measured (at least annually, more frequently if required to ensure sound management of nutrients). OR
You have developed a mass balance of nutrient production from your piggery or cattle feedlot using accepted design tools, such as PigBal, BeefBal or MEDLI using conservative figures. (There can be a great variation in nutrient predictions from mass balance models).

High Risk You have never measured, but only estimated the mass of nutrients applied using “text-book” values, such as those provided in Section 6.3.12 (piggeries) and 6.6.9 (feedlots) of the Resource Manual.

A risk weighting of 1 or 3 applies to the Nutrients and Manure criterion. A low risk attracts a risk weighting of “1” and high risk attracts a risk weighting of “3”.

Transfer these values to the Risk Assessment Matrices for soils, surface water and groundwater.

Since the mass of nutrients for reuse has only been estimated using text book values, the risk assessment of nutrients in manure and effluent is high, so a number of **3** can be applied.

Risk Assessing the Site

Transfer data from this assessment into the Design and Management Reuse Area Risk Assessment Summary table.

Size of Land Area

Knowledge of Size of Land Area

Low Risk From farm or paddock maps, you accurately know the area (ha) of each effluent or manure reuse paddock under each management regime (e.g. soil properties, land use).

Medium Risk You know the approximate area (ha) of each effluent or manure reuse paddock under each management regime.

High Risk You do not know the area of the effluent or manure reuse paddocks.

The area of the effluent irrigation area is approximately known from aerial photography, so the risk weighting is medium (2).

Knowledge of Yields of Crops or Pastures Grown on Reuse Areas

Low Risk	For your property and soil type, you know typical yields for the pastures or crops grown on reuse areas.
Medium Risk	You know typical district yields for the pastures or crops grown on reuse areas.
High Risk	You do not know typical yields for the pastures or crops grown on reuse areas.

For the property, typical dry matter yields are known, thus the risk weighting is low (**1**).

Knowledge of Nutrients Applied to Reuse Areas

Low Risk	You have calculated the nitrogen (kg/ha/yr) and phosphorus (kg/ha/yr) loading rates to reuse areas from estimated nutrient production.
High Risk	You have not calculated the nitrogen (kg/ha/yr) and phosphorus (kg/ha/yr) loading rates to reuse areas.

The nutrient application rate has not been calculated, thus the risk weighting is high (**3**).

Nitrogen Mass Balance for Reuse Areas

Low Risk	You have calculated that the net mass of nitrogen (kg/ha/yr) applied as effluent and / or solid by-products is exceeded by the mass of nitrogen (kg/ha/yr) that plant harvest should remove.
Medium Risk	You have calculated that the net mass of nitrogen applied (kg/ha/yr) as effluent or solid by-products is equal to the mass of nitrogen (kg/ha/yr) that plant harvest should remove.
High Risk	The net mass of nitrogen applied to reuse areas (kg/ha/yr) exceeds the mass removed <u>or</u> you do not know the net mass of nitrogen applied to the reuse area.

The net mass of nitrogen applied is not known, thus the risk weighting is high (**3**).

Phosphorus Mass Balance for Reuse Areas

Low Risk	You have calculated that the mass of phosphorus (kg/ha/yr) applied as effluent and / or solid by-products is exceeded by the mass of phosphorus (kg/ha/yr) that plant harvest should remove plus phosphorus storage calculated from a site-specific phosphorus sorption test.
Medium Risk	You have calculated that the net mass of phosphorus applied (kg/ha/yr) as effluent or solid by-products is equal to the mass of phosphorus (kg/ha/yr) that plant harvest should remove plus

phosphorus storage calculated from a site-specific phosphorus sorption test or from generic phosphorus sorption data for similar soil types.

High Risk — The net mass of phosphorus applied to reuse areas (kg/ha/yr) exceeds the mass removed plus storage calculated from a site-specific phosphorus sorption test or from generic phosphorus sorption data for similar soil types or you do not know the mass of phosphorus applied to the reuse area.

The net mass of phosphorus applied is not known, thus the risk weighting is high (**3**).

Using Appropriate Effluent & Solid By-Product Application Methods

If you reuse effluent on-site, select the appropriate risk category for "Effluent Irrigation" based on the information presented below. If you reuse solid by-products on-site, select the appropriate risk category for "Solids Spreading" from the information presented below. If you reuse effluent and solids on the same area, select the risk weighting that is highest from either the "Effluent Irrigation" or "Solids Spreading" sections below (e.g. if you have a rating of *low* for effluent irrigation and a rating of *medium* for solids spreading, the overall risk weighting you choose for the area is *medium*).

The results then need to be transferred into the "Using Appropriate Effluent & Solid By-Product Application Methods" row of Table 30 and converted into a risk weighting. A separate copy of Table 30 needs to be developed for separate reuse areas (e.g. effluent areas V solid areas) or reuse areas posing different risks (e.g. one effluent reuse area might be low risk, another high risk).

Effluent Irrigation

Low Risk — You use a low-pressure, travelling spray or drip irrigation system or a low-pressure solid set spray or drip irrigation system or a well designed and maintained flood irrigation system that is not on sandy to sandy loam soil. The system also applies effluent evenly and at target rates.

High Risk — You use a hand-shift sprinkler or hose or a poorly designed or managed flood irrigation system (e.g. land has not been levelled or effluent is unshandied or surface soil is sandy to sandy loam).

Effluent irrigation will be applied evenly with a low-pressure travelling irrigator, thus the risk weighting is low (**1**).

Solids Spreading

Low Risk — The spreading method used disperses solids evenly and at target rates.

Medium Risk — The spreading method used disperses solids fairly evenly and within 20% of target rates.

High Risk — The spreading method used disperses solids unevenly or at uncontrolled rates (not within 20% of target rates).

Table 67 is a template for summarising the design and management risk weightings for each design and management criterion. To complete the table, insert a risk weighting of 1, 2 or 3 against each criterion. A low risk attracts a risk weighting of "1", medium risk attracts a risk weighting of "2" and high risk attracts a risk weighting of "3". These numbers are transferred to Table 54, Table 72 and Table 73.

All solid manure will be sold off-farm.

TABLE 83 - DESIGN AND MANAGEMENT REUSE AREA RISK ASSESSMENT SUMMARY

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)
Size of land area	**3**
Application methods	**1**

Also transfer these values to the Risk Assessment Matrices for soils, surface water and groundwater.

Soil

Texture

Low vulnerability: soil texture is loam to medium clay.

Medium vulnerability: soil texture is duplex with a light topsoil and a heavy subsoil or is heavy clay.

High vulnerability: soil texture is sand or unknown.

The soil texture of the effluent irrigation area is sandy-loam, thus the risk weighting is medium (**2**).

Depth

Low vulnerability: Depth of soil is > 1 m.

Medium vulnerability: Depth of soil is 0.5 – 1m.

High vulnerability: Depth of soil is < 0.5 m or unknown.

The soil depth of the effluent irrigation area is at least 1 m, thus the risk weighting is low (**1**).

Slope

Low vulnerability: Slope is < 5% or slope is 5-10% but continuous vegetative cover is constantly maintained over the area or slope is 5-10% but a system of well-designed contour banks is in place to slow the movement of water from the site.

Medium vulnerability: Slope is 5 – 10% or slope is >10% but continuous vegetative cover is constantly maintained over the area or slope is >10% but a system of well-designed contour banks is in place to slow the movement of water from the site.

High vulnerability: Slope is > 10% or unknown.

The effluent irrigation area is flat, thus the risk weighting is low (**1**).

Soil Dispersion

Low vulnerability: Soil does not disperse on wetting and has a low exchangeable sodium percentage (less than 6%).

Medium vulnerability: Soil disperses on wetting and / or has an exchangeable sodium percentage of 6-15%.

High vulnerability: Soil disperses on wetting and / or has an exchangeable sodium percentage exceeding 15% or the dispersive behaviour and exchangeable sodium percentage of the soil are unknown.

The exchangeable sodium percentage of the topsoil is 10%, thus the risk weighting is medium (**2**).

Salinity

Low vulnerability: Soil is in the very low to low salinity class (EC_{se} is less than 1.9 dS/m)

Medium vulnerability: Soil is in the medium salinity class (EC_{se} is 1.9-4.5 dS/m)

High vulnerability: Soil is in the high to extreme salinity class (EC_{se} is over 4.5 dS/m) or soil salinity class is unknown.

The soil has a low to very low salinity class to depths of 60 cm, thus the risk weight is low (**1**).

Nutrient Status

Nitrogen

Low vulnerability: Either soil solution nitrate-N levels at the base of the active root zone are <10 mg/L or are less than measured baseline data.

High vulnerability: Either soil solution nitrate-N levels at the base of the active root zone are >10 mg/L or are greater than measured baseline data.

These can be converted to soil nitrate-nitrogen concentrations for different soil types as per Table 31 (Section 8.1.7) of the Resource Manual.

Nitrate-N concentrations measured to a depth of 60 cm exceeds the recommended maximum 1.5 mg/kg of nitrate-N at the base of the root zone for a sandy-loam, thus the risk is high (**3**).

Phosphorus

Vulnerability ratings for phosphorus are based on three methods.

Method 1 involves a check as to whether the Colwell Extractable phosphorus levels exceed certain limits. These limits are based on measured Colwell extractable phosphorus for numerous soils (categorised by clay content and pH). The upper limits (high rating) are one standard deviation above the mean of numerous Colwell extractable phosphorus levels (Redding, 2002). However, these limits may not be appropriate for some soil types, such as black vertosols, which may have high levels of Colwell phosphorus in their 'virgin' state.

Method 1 (Most Soils)

Low vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	< 15
< 30%	> 7	< 30
> 30%	< 7	< 40
> 30%	> 7	< 45

Medium vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	15 – 30
< 30%	> 7	30 – 60
> 30%	< 7	40 – 75
> 30%	> 7	45 – 85

High vulnerability:

Clay Content	Soil pH	Colwell Extractable phosphorus Level (mg/kg)
< 30%	< 7	> 30
< 30%	> 7	> 60
> 30%	< 7	> 75
> 30%	> 7	> 85

Colwell extractable P levels have been measured at 17 mg/kg in the top 10 cm. The topsoil has a pH of 5.8 and we can assume the clay content is less than 30%, thus the risk weighting is medium (**2**).

Surface Water

Surface water includes water in dams, reservoirs, rivers, creeks and all other waterways where rainfall is likely to collect. Ideally, reuse areas should be well separated from surface water bodies, particularly those used for sensitive purposes e.g. town water supplies. However, distance is not the only criterion determining the potential for contamination from reuse areas. Design and management factors, particularly the amount and type of vegetative cover, may significantly reduce any potential contamination of surface waters.

Water Quality Protection

Low vulnerability: Reuse area is at least 200 m from a surface water body and effluent irrigations do not cause runoff <u>or</u> is at least 150 m from a surface water body but includes a vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff <u>or</u> is at least 100 m from a surface water body but includes a well-maintained vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff <u>or</u> there is a terminal pond sized to catch the first 12 mm of rainfall runoff plus irrigation water runoff.

Medium vulnerability: Reuse area is between 100 m and 200 m from a surface water body and effluent irrigations do not cause runoff <u>or</u> is at least 75 m from a surface water body but includes a vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff <u>or</u> is at least 50 m from a surface water body but includes a well-maintained vegetative buffer at least 25 m wide and effluent irrigations do not cause runoff.

High vulnerability: Reuse area has no vegetative buffer and is less than 100 m from a surface water body <u>or</u> reuse area has a vegetative buffer but is within 50 m of a surface water body <u>or</u> and effluent irrigations create runoff that is not captured in a terminal pond.

The effluent irrigation area is 100 m from the creek and effluent reuse does not cause runoff. The surface water protection measures in place match the risk weighting of medium (**2**).

Flood potential

Low vulnerability: reuse area is above the 1 in 10 year flood line.

Medium vulnerability: reuse area is above the 1 in 5 year flood line but below the 1 in 10 year flood line.

High vulnerability: reuse area is below the 1 in 5 year flood line <u>or</u> flooding frequency of reuse area is unknown.

The flood frequency is one in ten years, thus the risk weighting is medium (**2**).

Groundwater

Ideally, reuse areas should be located on areas with deep groundwater or on those well protected by a clay blanket or confined aquifer. The risk to groundwater from effluent reuse depends upon the protection afforded by soil type (e.g. a deep clay blanket may afford good protection, a sandy loam soil provides relatively poor protection) and the geology and type of aquifer (e.g a confined aquifer versus an alluvial aquifer).

The consequences of nutrient or salt leaching to groundwater depend on the quality of the groundwater (e.g. potable water V brackish water).

Depth to groundwater

Low vulnerability: Groundwater is at least 20 m below the surface, or there is no significant groundwater.

Medium vulnerability: Groundwater is 10-20 m below the surface.

High vulnerability: Groundwater is less than 10 m below the surface or depth to groundwater is unknown.

Depth to groundwater is known to be about 50-60 m deep, thus the risk weighting is low (**1**).

Soil type

Low vulnerability: There is at least 0.5 m of clay above the aquifer or the aquifer is confined, or there is no significant groundwater.

Medium vulnerability: There is at least a metre of loam to clay soil above the aquifer.

High vulnerability: The soil above the aquifer is sand, sandy-loam or gravel or there is less than a metre of soil above the aquifer or the soil type above the aquifer is unknown.

There is deep clay under the sandy-loam soil, thus the risk weighting is low (**1**).

Water quality

Low vulnerability: The groundwater resources in the area are of a quality having no productive use e.g. EC exceeds 8 dS/m, or there is no significant groundwater.

Medium vulnerability: Groundwater resources are suitable for stock drinking water and irrigation e.g. EC of up to 8 dS/m & containing less than 100 mg NO_3N/L

High vulnerability: Groundwater resources are suitable for human consumption. (EC of up to 1.6 dS/m and containing less than 10 mg NO_3N/L) or the quality of groundwater resources is unknown.

Tests of bores suggest that the bore water in the area is suitable for stock consumption, but not suitable for human consumption. Thus the risk weighting is medium (**2**).

Table 68, Table 69 and Table 53 are templates for recording the site vulnerability risk weightings for soil, surface water and groundwater. To complete the tables, a vulnerability weighting of 1, 2 or 3 applies to each sub-category of soil, surface water and groundwater. A low vulnerability attracts a vulnerability weighting of “1”, medium vulnerability attracts a vulnerability weighting of “2” and high vulnerability attracts a vulnerability weighting of “3”. These numbers are transferred to Table 54, Table 72 and Table 73.

Risk Assessment Tables

TABLE 84 - VULNERABILITY WEIGHTINGS - SOIL

Resource	Texture (weighting low = 1, med. = 2, high = 3)	Depth (weighting low = 1, med. = 2, high = 3)	Slope (weighting low = 1, med. = 2, high = 3)	Soil Dispersion (weighting low = 1, med. = 2, high = 3)	Salinity (weighting low = 1, med. = 2, high = 3)	Nitrogen (weighting low = 1, med. = 2, high = 3)	Phosphorus (weighting low = 1, med. = 2, high = 3)
Site Vulnerability Weighting	**1**	**1**	**1**	**2**	**1**	**3**	**2**

TABLE 85 – VULNERABILITY WEIGHTINGS – SURFACE WATER

Resource	Water Quality Protection Weighting (low = 1, medium = 2, high = 3)	Flood Potential Weighting (low = 1, medium = 2, high = 3)
Site Vulnerability Weighting	**2**	**2**

TABLE 86 – VULNERABILITY WEIGHTINGS - GROUNDWATER

Resource	Depth to Groundwater Weighting (low = 1, medium = 2, high = 3)	Soil Type Weighting (low = 1, medium = 2, high = 3)	Water Quality Weighting (low = 1, medium = 2, high = 3)
Site Vulnerability Weighting	**1**	**1**	**2**

Transfer these values to the Risk Assessment Matrix Tables.

TABLE 87 - RISK ASSESSMENT MATRIX - SOIL

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Texture	Depth	Slope	Soil Dispersion	Salinity	Nitrogen	Phosphorus
		2	**1**	**1**	**2**	**1**	**3**	**2**
Nutrients in manure and effluent	**3**	**6**	**3**	**3**	**6**	**3**	**9**	**6**
Size of land area	**2**	**4**	**2**	**2**	**4**	**2**	**6**	**4**
Application method	**1**	**2**	**1**	**1**	**2**	**1**	**3**	**2**

TABLE 88 - RISK ASSESSMENT MATRIX – SURFACE WATER

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Site Vulnerability Weighting	
		Water Quality Protection	Flood Potential
		2	**2**
Nutrients in manure and effluent	**3**	**6**	**6**
Size of land area	**2**	**4**	**4**
Application method	**1**	**2**	**2**

TABLE 89 - RISK ASSESSMENT MATRIX - GROUNDWATER

Design and Management Criteria	Design & Management Risk Weighting (Low = 1, medium = 2, high = 3)	Site Vulnerability Weighting		
		Depth	Soil Type	Water Use
		1	**1**	**2**
Nutrients in manure and effluent	**3**	**3**	**3**	**6**
Size of land area	**2**	**2**	**2**	**4**
Application methods	**1**	**1**	**1**	**2**

Monitoring Based on Risk

Ratings of 1 and 2 require minimal monitoring. Ratings of 3, 4 & 6 attract moderate levels of monitoring. A rating of 9 requires intensive monitoring. It is important to realise that if a rating of 4 is calculated for groundwater and a rating of 9 is calculated for soil, moderate monitoring would be warranted for the groundwater and intensive monitoring would be warranted for the soil.

Soils

Where the risk of soil related impacts is low (rating of 1-3) and at least 3 years of annual monitoring shows that the system is sustainable, it is suggested that soils from reuse areas should be monitored at least every three years. Those in a low risk category will not need to monitor effluent quality *unless* they are already undertaking this monitoring (which is the reason for being in this category).

Where there is a medium risk of soil impacts (rating of 4 or 6) and at least 3 years of monitoring data show that the system is sustainable, it is suggested that soils from reuse areas should be sampled and analysed at least every two years. Effluent and solids quality (if reused on-site) should also be analysed annually.

Where there is a high risk of soil impacts (rating of 9), annual soil monitoring is imperative. Effluent and solids quality (if reused on-site) should also be analysed annually.

Surface Water

Surface water quality monitoring is not suggested as a relevant measure of sustainability for piggeries and cattle feedlots, as they are not direct discharge industries (e.g. sewage treatment plants) and generally rely on land application for the reuse of by-products. To be able to achieve any meaningful results from a monitoring perspective, surface water monitoring would require sophisticated equipment and trained operators.

Piggeries and cattle feedlots are required to comply with relevant codes of practice for their design and management, such as appropriate buffers, vegetative filter strips or terminal ponds. If an enterprise attracts a high rating, remedial action in the form of improved design and/or management of the reuse are would be required.

Groundwater

Groundwater quality monitoring would be warranted for anyone attracting a high rating (9). Ideally this would include sampling and analysis from bores upslope and downslope of reuse areas. Electrical conductivity and nitrate-nitrogen should be determined. On very sandy soils, total P should also be measured. If a moderate risk weighting is attracted for groundwater, monitoring would not necessarily be required, provided nutrient and salt risk weightings for the soil are low.

Soil monitoring

The elevated nutrient and ESP content of the soil coupled with an absence of knowledge about the nutrients in the effluent and the exact size of the reuse area necessitates intensive monitoring. There is potential to immediately reduce the required frequency of monitoring by quantifying the nutrients for reuse using mass balance principles.

Surface water monitoring

The surface water risk assessment produced moderate levels of risk, due to the combination of lack of knowledge about the nutrient content of the effluent coupled with the proximity of the reuse area to the creek and the flooding potential of the site. Again, quantifying the nutrients for reuse may significantly reduce the assessed risk.

Groundwater monitoring

The groundwater risk assessment produced moderate levels of risk, due mainly to the reasonably high quality of the groundwater and uncertainty about the quantity of nutrients for reuse. Again, the assessed risk could potentially be reduced by accurately knowing the mass of nutrients for reuse.