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Effluent and Manure Management Database for the Australian Dairy Industry

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ii. Purpose

Dairy effluent is described as a potential point source of nutrient pollution for waterways and mismanagement of it risks impacting the environment. Environmental legislation in all Australian States and Territories has set a minimum standard that dairies must comply with to prevent nutrient pollution leaving the farm boundaries. Compliance with such legislation requires that the dairy industry have access to up-to-date and validated technical information about options for effectively managing its effluent.

The Australian dairy industry has changed dramatically since deregulation in 2000, with increased average herd size and increased intensity of use in supporting areas around the dairy. The increased volumes of effluent and solids generated must be considered in effluent and manure management plans.

The dairy industry is truly a national industry with most dairy companies sourcing milk from more than one State. Although recommendations for best practices for effluent management within each State will be influenced by regulatory requirements, recommendations on the technical aspects of effluent management for dairy farmers are applicable across State borders.

The Effluent and Manure Management Database for the Australian Dairy Industry is a repository of reliable and scientifically validated technical information on dairy effluent management adaptable to all dairying regions in Australia. The database outlines the principles for effective effluent management, performance based design criteria for components of effluent containment and reuse systems, and appropriate management principles for optimal operation of each design.

The database not only provides the technical information required for on farm effluent management designs but also the technical base to support National, State and Regional regulations on dairy effluent management, technical and farmer based extension programs, and educational material on dairy effluent management.

The audience for the database is primarily persons and groups who give advice to farmers, produce extension material for farmers, or design equipment for effluent management, and as a dairy specific technical basis for regulation. While the information presented is therefore by design technical in nature, not explanatory, it is expected that some farmers will also be able to use the database.

To ensure that the information presented is valid and relevant, the database has been reviewed by a committee of technical experts representing each State. For the benefit of the user, where the information presented can be referenced, a hyperlink to the database entry is provided at its first citation; subsequent citations in a chapter are not hyperlinked to avoid unnecessary clutter. The database entry will, where possible, have an additional hyperlink to any associated internet content.

Finally, as new research and information is continually being developed, it is intended that the Effluent and Manure Management Database for the Australian Dairy Industry will be up-dated on a regular basis.

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1.1 Physical, biological and chemical components of effluent and manure

Terminology

Collectively, urine and dung are called **excreta**. This excreta is typically mixed with wash water produced by cleaning yards; with wash water, chemicals and residual milk from cleaning equipment; with waste feed or bedding material; and occasionally with rainwater. The resulting liquid is usually referred to as **effluent** (or dairy shed effluent or wastewater).

Excreta that dries before being collected (for example, by scraping from feedpads or loafing yards) and is handled as a semi-solid or solid is called **manure**. Manure can also contain waste feed or bedding material and soil removed by scraping non-concrete areas.

The characteristics of effluent and manure need to be understood before a suitable option for management can be selected. The relevant characteristics can be described by the following physical, biological and chemical parameters.

Physical—solids

The solids in effluent and manure can be partitioned into different physical components, as described by the following matrix adapted from Taiganides (1977):

$$\begin{array}{ccccc} \text{TS} & = & \text{VS} & + & \text{FS} \\ \parallel & & \parallel & & \parallel \\ \text{SS} & = & \text{VSS} & + & \text{FSS} \\ + & & + & & + \\ \text{TDS} & = & \text{VDS} & + & \text{FDS} \end{array}$$

where TS = total solids
VS = (total) volatile solids
FS = (total) fixed solids
SS = (total) suspended solids
VSS = volatile suspended solids
FSS = fixed suspended solids
TDS = total dissolved solids
VDS = volatile dissolved solids
FDS = fixed dissolved solids.

The characteristics of effluent and manure can be described by these components and other biological and chemical parameters, as explained below.

Total solids (TS)

The choice of effluent management system is constrained by the total solids content of the material to be handled. Figure 1 shows generally accepted TS limitations for different manure handling options.

The TS content of manure ‘as-excreted’ may range from 8% to 15% and can therefore be described as a liquid or semi-liquid (a slurry). Material of this concentration is usually conveyed by augers or manure tankers. After yard and plant wash is added, the TS content of the diluted effluent is usually between 0.5% and 1.2% (Longhurst *et al.* 2000).

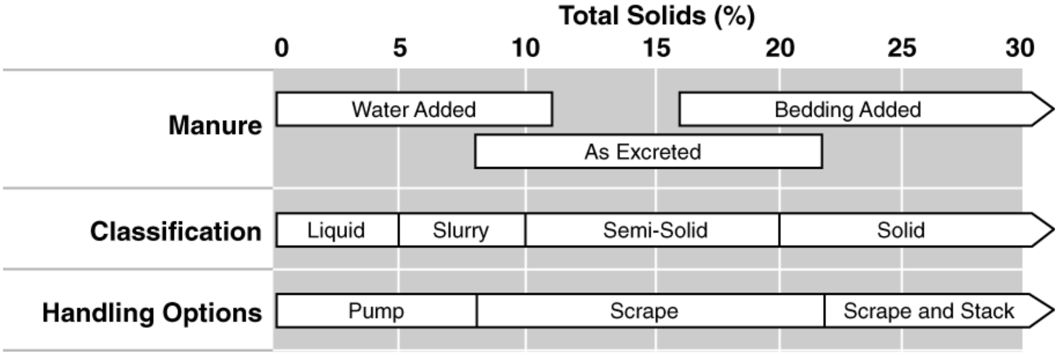

Figure 1. TS and manure handling options (USDA-NRCS 1996)**.**

Note that a TS concentration of 10 000 mg·L^{-1} (or 10 g·L^{-1}) is equivalent to 1% solids; the density of manure is almost the same as that of water up to around 10% TS (Taiganides 1977).

Volatile solids (VS)

The volatile solids component is the organic matter or degradable component that must be removed or stabilised during treatment. The VS component of dairy cattle faeces is generally 80% to 86% of TS, the remainder being ash (FS) (Zhang *et al.* 2003, Wright 2005, ASAE 2005). Any extraneous material such as laneway material walked in on hooves, soil washed from earthen pads or sand bedding entering the effluent stream will reduce the ratio of VS to FS.

Fixed solids (FS)

The fixed solids constitute the residual inorganic compounds (N, P, K, Ca, Cu, Zn, Fe etc.) in a suspended or dissolved state. In dilute effluents, these minerals are mainly dissolved, and their removal from the effluent stream is difficult.

Suspended solids (SS)

The content of total SS ranges from 62% to 83% of TS Loehr (1984), and sets the theoretical limit of performance for separation systems (see chapter 2.1 Solid-liquid separation systems). The majority of SS is volatile (VSS): approximately 80% according to Longhurst *et al*. (2000); the rest is fixed (FSS).

Total dissolved solids (TDS)

All dissolved solids (TDS) are ions. There is a strong correlation between TDS and the electrical conductivity (EC) of effluent.

Biological oxygen demand (BOD)

Biological oxygen demand is an index of the oxygen-demanding properties of biodegradable material in water. It is a useful measure for assessing the strength of effluent and its pollution potential. The BOD curve in Figure 2 illustrates the typical two-stage characteristic: the first stage is related to demand for carbon, and the second to nitrification. Because the reproductive rate of the bacteria responsible for nitrification oxygen demand is slow, it normally takes 6 to 10 days for them to influence the BOD measure (Metcalf & Eddy Inc. 2003).

Unless specified otherwise, BOD values usually refer to the standard 5-day value (BOD_5), measured within the carbon demand stage. Note that the BOD_5 of animal effluents cannot be compared to that of sewage, as BOD_5 of sewage represents 68% to

80% of the ultimate BOD, whereas that of animal effluents is only 16% to 26%. (Having already undergone anaerobic fermentation in the rumen, the effluent contains a higher proportion of slowly degradable organic matter.)

Typically, dairy effluent (unless substantially diluted) has a BOD_5 of the order of 2500–4000 mg·L^{-1}. Although much of the organic matter in dairy effluent is derived from manure, the contribution from spilt milk or flushing milk lines cannot be ignored. Raw milk has a BOD_5 of 100 000 mg·L^{-1} and has the potential to be a powerful pollutant if inappropriately managed.

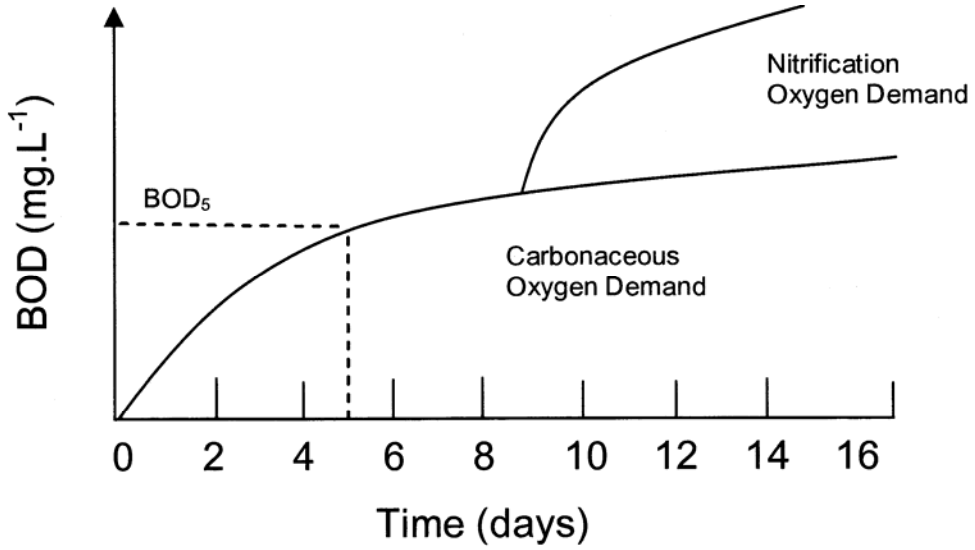

Figure 2. Typical BOD response with time.

Chemical oxygen demand (COD)

Chemical oxygen demand is the amount of oxygen consumed during the oxidation of organic carbon under a high-temperature, strongly acidic chemical digestion process. COD is frequently used in monitoring treatment processes, as it can be completed in 1 to 3 h (rather than the 5 days for BOD_5). However, since it is a chemical process, the biodegradability prospects for the material are not given.

The COD:BOD_5 ratio is frequently used as an indicator of biological degradability: ratios exceeding 5:1 indicate low digestibility. The COD:BOD_5 ratio of dairy effluent is typically 7:1 to 12:1.

Nutrient content and distribution by manure particle size

Particle size distribution (PSD) is important when we are considering nutrient balances and the impact of separation systems (see chapter 2.1 ‘Solid-liquid separation systems’). The limited data regarding particle size distribution of dairy manure comes mainly from ration-fed animals in the USA. Meyer *et al*. (2007) reported the results of a PSD study of four lactating cows fed a diet of coarsely chopped lucerne, whole cottonseed and a concentrate mix (intake averaged 21.9 kg DM day^{-1} with a nutrient composition of 2.7% N, 0.4% P and 1.5% K). It is expected that the distribution of particle sizes in manure will vary with diet.

Meyer *et al*. (2007) identified that around half of the mass and most of the nutrients in fresh dairy manure are associated with particle sizes smaller than 125 µm: 50% of TS, 85% of N, 87% of P and 99.8% of K. Just 37% of TS, 10% of N and P and negligible K were associated with particles larger than 1000 µm—a size range that may be removable by mechanical screens. Wright (2005) reported similar results.

Table 2. Particle size distribution for 'as-excreted' dairy manure (Meyer et al. 2007).

Particle size (µm)	Percentage of total solids	Percentage of N	Percentage of P	Percentage of K
2000	30	9.0	9.0	0.1
1000	7	1.5	0.8	0.0
500	6	1.6	1.1	0.0
250	5	1.7	1.4	0.0
125	3	1.4	1.4	0.0
<125	50	84.9	86.7	99.8

Meyer *et al*. (2007) confirmed the commonly assumed partitioning of nutrients between faeces and urine from dairy cattle: N, 48% faecal, 52% urine; P, 97% faecal, 3% urine; K, 30% faecal, 70% urine.

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1.2 Characteristics of effluent and manure

An understanding of the characteristics and quantity of effluent and manure generated by the cow is the starting point for assessing the suitability of effluent and manure management strategies for a dairy farm. The nature of excreta produced by the cow depends on breed, dry matter intake and composition of the diet. The nature of the effluent generated varies much more, depending on the number of hours on a cleaned surface, type of washdown systems, size of catchment area and climate. For this reason, it is preferable to work with estimates of the mass of 'as-excreted' manure (faeces + urine) and its components rather than to rely on 'typical' effluent analysis data.

Rates of faeces and urine production

Many guidelines have used data from the ASAE Standard 'Manure Production and Characteristics' (ASAE 1999) to characterise manure (a term usually taken to include both faeces and urine). That standard was based on 1960s and 1970s US data, so it presumably contained little, if any, data from grazing operations. Strategies for feeding cows in Australia, however, are diverse: from grazing with minimal supplementary feeding, through the broad group of grazing operations with varying degrees of increased supplementary feeding on feedpads, to completely ration-fed cows in freestalls (where the cows have a more predictable manure production). When the majority of the feed intake is sourced from pasture, variation in the forage consumed, both spatially and temporally, makes estimation of volume and composition difficult.

It is widely documented that milk production depends on feed intake: high-producing cows have a larger dry matter intake and consequently a larger volume of excreted manure than lower-producing cows. Nennich *et al*. (2003) found that milk production was a better indicator of the mass of faeces produced, and its constituents, than body weight; that work forms the basis of the updated ASAE Standard (ASAE 2005). Nennich et al. (2005) went on to define regression equations for estimating manure production based on either milk yield or dry matter intake and dietary concentrations. Although the latter were more accurate than milk yield alone, the difficulty in determining those parameters for grazing operations restricts their use to more intensive operations.

Table 1 compares the manure production predicted by the sources discussed above with the little data available from Australian and New Zealand sources. It is clear that there will be considerable variation in the volume and characteristics of manure produced from pasture-based operations, but little published research documents that range. In the absence of any more relevant data, the equations developed by Nennich *et al*. (2005) are the most useful tool available and may be adopted unless site-specific information is available.

Equations based on milk yield developed by Nennich *et al*. (2005):

$$\text{Total excreta (faeces + urine) (kg}\cdot\text{day}^{-1}) = [\text{milk (kg}\cdot\text{day}^{-1}) \times 0.616] + 46.2 \qquad (1)$$

$$\text{Dry matter excretion (kg}\cdot\text{day}^{-1}) = [\text{milk (kg}\cdot\text{day}^{-1}) \times 0.0874] + 5.6 \qquad (2)$$

$$\text{N excretion (g}\cdot\text{day}^{-1}) = [\text{milk (kg}\cdot\text{day}^{-1}) \times 2.82] + 346 \qquad (3)$$

$$\text{P excretion (g}\cdot\text{day}^{-1}) = [\text{milk (kg}\cdot\text{day}^{-1}) \times 0.781] + 50.4 \qquad (4)$$

$$\text{K excretion (g}\cdot\text{day}^{-1}) = [\text{milk (kg}\cdot\text{day}^{-1}) \times 1.476] + 154.1 \qquad (5)$$

1.2 Characteristics of effluent and manure

Table 1. Comparing estimated manure production.

	ASAE (1999)	ASAE (2005)	Nennich *et al*. (2005)	Vanderholm (1984)	Victoria [d]	Victoria [d]
Relevance	USA	USA	USA	NZ (pasture)	Victoria (pasture + grain)	Victoria (pasture + protein)
Milk yield (kg·day^{-1})		*16.5*	*16.5*			
Body weight (kg)	*600*	*600*		*600*		
Total manure (faeces + urine) (kg·day^{-1})	52	54	56	65		
Urine (kg·day^{-1})	16	22		30	10–18	15–31
TS (kg·day^{-1})	7.2	6.6	7.0	5.3	3–5	4–6.6
VS (kg·day^{-1})	6.0	5.5 [a]	5.9 [a]	3.8		
N (g·day^{-1})	270	351	393	290	265 (113 dung, 152 urine)	394 (159 dung, 235 urine)
P (g·day^{-1})	56	58	63	30	48 (42 dung, 6 urine)	62 (59 dung, 3 urine)
K (g·day^{-1})	174	60	178	370	161 (30 dung, 131 urine)	203 (19 dung, 184 urine)
BOD_5 (kg·day^{-1})	1.0	0.9 [b]	0.9 [b]	1.2		
COD (kg·day^{-1})	6.6	6.1 [c]	6.5 [c]	5.2		
Moisture content (incl. urine) (%)	13.8	12.2	12.5	8.1		

[a] VS calculated as 0.83 × TS (ASAE 1999)
[b] BOD calculated as 0.16 × VS (ASAE 1999)
[c] COD calculated as 1.1 × VS (ASAE 1999)
[d] Personal communication, D. Daley 2003, DPI Ellinbank.
Note that numbers in italics are selected for enabling a comparison between sources. The milk yield of 16.5 L·day^{-1} was based on the estimated 2005–06 Australian average annual production of 5034 L (Dairy Australia 2006) and a 305-day lactation.

Minimising excreted nutrients

Powell (2006) suggests that ‘manure management should start at the front, rather than the back end of the animal.’ Dietary nutrient content in excess of requirements is excreted: excess protein (as urea) and K in urine, excess P in faeces. Adding excess salt to stimulate appetite can exacerbate problems with salinity management for the reuse area. Although the opportunity to manipulate pasture-based diets is limited and recommendations for minimum nutrient levels are beyond the scope of this manual, the issue is important when you are considering both whole-farm nutrient balances and opportunities for waste minimisation. In general, the formulation of supplementary feeds should be based on nutritional requirements to avoid overfeeding and thus to reduce the excretion of undigested components.

Proportion of daily manure output collected

Most existing guidelines assume that 10% to 15% of the daily manure output (equivalent to 2.4 to 3.6 h per day) is deposited onto surfaces from which effluent is collected. Although that is a reasonable estimate for the holding yard at the dairy, industry trends towards feeding increasing levels of supplements or mixed rations mean that such an assumption is no longer valid. On a farm with a feedpad, such an assumption can seriously underestimate the volume of manure and nutrients to be handled. The amount of time that cows are confined to a collected surface must be determined on each farm.

Dunging behaviour

As a general rule, the amount of manure deposited at any location is proportional to the time that the cows spend at that location. Although the frequency of defecation and urination increases after the cow rises from a resting or rumination period, White *et al*. (2001) found that when cows were given time to void themselves before being retrieved for milking, the volume of faeces and urine deposited at the dairy was proportional to the time that the cows were held.

As the amount of manure to be collected can be minimised by reducing the time spent on a confined surface, consider opportunities to improve dairy throughput and not hold cows after milking when planning or reviewing an effluent management system. To reduce the amount of manure collected in laneways, give cows sufficient time to stand and defecate in the paddock before being moved. Stress is one reason for an increase in the number of defecations and urinations at the dairy; avoid rough handing and crowding in holding yards. More detailed information on shed and yard configuration and its impact on throughput is provided by the CowTime program (http://www.cowtime.com.au/).

Time in the yard

The average time cows were held for milking ranged from 2.8 h per day until the last cow left the yard (40–100 cows) to 3.8 h per day (>150 cows), with an overall mean of 3.3 h per day (Wheeler 1996). Where cows are allowed to return to the paddock after milking, the time that the middle cow spends at the yard is more appropriate for calculations; Wheeler's results suggested a range of means from 1.6 to 2.1 h, with an overall mean of 1.8 h per day. Therefore, the rule-of-thumb for the proportion of manure collected from the holding yard should be 10% where cows return to the paddock immediately after milking, and 15% where cows are held until milking is complete.

Impact of supplementary feedpads on manure proportion

The adoption of feedpads to support more intensive feeding strategies and to alleviate environmental stresses significantly increases the time cows are restricted to a surface from which manure must be collected (see chapter 4.2 'Feedpads, calving pads and loafing pads'). It is no longer uncommon for cows to be on or around a feedpad for 8 to 12 h per day, particularly where it is covered.

Although the volume of manure collected will increase significantly under such condition, some assumptions need to made regarding its distribution around the feedpad. Dairy cows will eat for up to 8 h a day. Therefore, if given free access to a loafing area, cows fed on an uncovered feedpad may deposit one-third of the manure on the feedpad and the remainder on the loafing area (which may be handled separately as a scraped solid). If the feedpad is covered, the proportion collected may well be higher, particularly in summer.

Any summary of the design criteria for an effluent management system (the composite of the collection, storage and/or treatment, and reuse components) should clearly report the likely times when the herd is confined and the proportion of daily output to be collected that was adopted in developing the plan.

Water use in dairies

Most of the water used in and around the dairy ends up being collected by the effluent management system. An accurate estimate of the water usage by the dairy is needed in order to successfully design and manage an effluent system: not only to ensure sufficient storage (see chapter 2.6 'Effluent storage requirement'), but also to secure a supply with sufficient quality and quantity, and to identify opportunities for minimisation.

1.2 Characteristics of effluent and manure

There is no simple relationship between the amount of water used for cleaning yards and the number of cows milked or the area of the yard (Wheeler 1996). McDonald (2005) found that total water use in dairies can vary significantly between 1000 L and over 150 000 L day^{-1}, and that it is not necessarily the larger operations and dairy sheds that use excessive amounts of water.

Typical ranges in water use reported include 11 to 56 L day^{-1} per cow (NSW Dairy Effluent Subcommittee 1999) and 4 to 138 L day^{-1} per cow (Rogers and Alexander 2000). The survey of 114 Bonlac farms by Rogers and Alexander (2000) recorded an average water use of 33 L·day^{-1} per cow (Table 3).

Table 3. Reported water use (Rogers and Alexander 2000).

	Average (*n* = 114) L·day^{-1} per cow	Rotary L·day^{-1} per cow	Herringbone L·day^{-1} per cow
Yard	21	26	19
Pit	4.4		
Platform	1.3	10	n.a.
Cups	1.4		
Teats	0.1	n.a.	n.a.
Cold machine wash	1.6	1.9	1.6
Hot machine wash	2.0	2.3	1.7
Vat	1.0	n.a.	n.a.
Total	33	47	27

Water use audits

Producers tend to underestimate daily water use, so a thorough audit is required for an accurate estimate. A water use audit should include at least the following:

- yard washing (distinguishing between clean water and reuse of treated effluent)
- yard pre-wetting
- pit or platform wash
- cup sprays
- platform sprays
- teat wash
- milking machine wash
- vat wash
- platecooler
- cow cooling.

Planning for new dairies without an audit

Although it is preferable to base effluent volumes on data from the actual farm, planning for a new site often requires that 'typical' data be used, as an audit cannot be done. McDonald (2005) suggests that in the absence of any other data, reasonable estimates for water use can be based on the number of milking units (Figure 1).

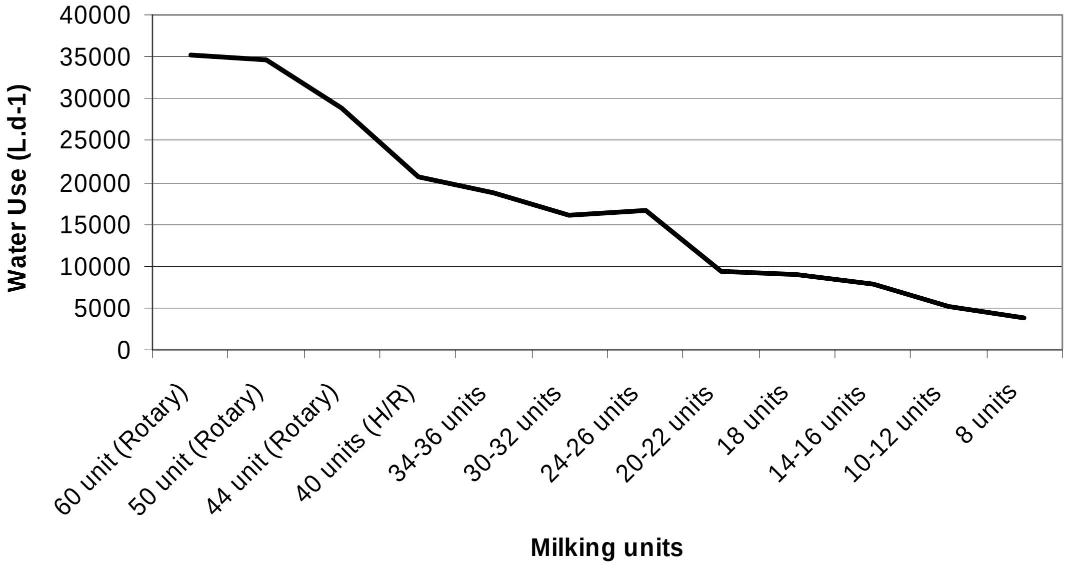

Figure 1. Total water use per day versus shed or milking units (McDonald 2005).

Minimising water use

The range in water use recorded by McDonald (2005) suggests that significant savings are possible without compromising plant hygiene or milk quality. For example, cup and platform sprays in rotary sheds represent up to 40% of total water use, but this component could be reduced by 70% to 80%.

To minimise the volume of effluent generated, Rural Solutions SA (2005) recommend reducing the cup spray flow rate to 10–15 $L{\cdot}min^{-1}$ for most of the milking; the fine spray is sufficient to prevent manure from sticking to the cups. On the last rotation, the flow rate can be turned on full to 70 to 80 $L{\cdot}min^{-1}$ for a final cup wash. They also recommend that the platform spray be used strategically (when there is manure that must be removed), but otherwise left off until the last rotation.

Water use for washing milking plant and vats should not be reduced below manufacturers' recommendations.

Floodwash systems represent the largest use of water around the dairy; where possible they should draw on treated effluent for supply and not clean water. Even if treated effluent is used, water use should be examined, as McDonald (2005) suggests that many farmers are dumping the entire tank volume when it may not be necessary. Shutting off the flow after the required flush duration will improve the performance of the solids trap and reduce the volume to be handled by any pumps. The volume of water required should be assessed by using appropriate flow depth, velocity and duration criteria (see chapter 1.4 'Floodwash systems').

Platecooler water use is 2.5 to 3 times the volume of milk cooled. As this water remains clean, it should be reused for washing or stock consumption. If excess platecooler water is generated, it should be directed to the effluent system only if its volume is allowed for in the calculation of storage requirement (see chapter 2.6 'Effluent storage requirement').

Although the volume of water used for cow cooling can be significant, the water is generally not needed during the storage period and should not affect the storage requirement.

Stormwater

Contaminated stormwater must be collected and treated, but diverting clean stormwater away from the effluent collection point will reduce the volume that the effluent management system must handle, decreasing the size of the storage pond and reducing the volume that has to be disposed of by irrigation. See chapter 2.6 ‘Effluent storage requirement’ for further information on stormwater minimisation and yard runoff diversion.

In summary, all dairies should:

- collect and use roof runoff by directing gutters to the tank supplying platecooler or washdown water
- reuse platecooler discharge for washdown
- prevent runoff from entering the yards or effluent systems from upslope.

Typical raw effluent analysis

A table of ‘typical’ raw effluent analyses is provided for background information. For the reasons stated above, typical concentrations should not to used for design purposes unless site-specific information is not available.

Table 4. Typical dairy shed effluent concentrations.

Parameter	Units	Region	
		NZ [a] (*n* > 37)	Victoria (NE) [b]
Total solids (TS)	%	0.9 (typical range 0.5–1.2)	
BOD_5	$mg{\cdot}L^{-1}$		3200
Suspended solids (SS)	$mg{\cdot}L^{-1}$		2400
Total N	$mg{\cdot}L^{-1}$	269	187
Organic N	$mg{\cdot}L^{-1}$	219	
Ammonium + ammonia	$mg{\cdot}L^{-1}$	48	84
Total P	$mg{\cdot}L^{-1}$	69	26
K	$mg{\cdot}L^{-1}$	370	200
Total S	$mg{\cdot}L^{-1}$	65	
Electrical conductivity (EC)	$dS{\cdot}m^{-1}$		1.12
pH	–		8.0
Cl^-	$mg{\cdot}L^{-1}$		180
Mg^{2+}	$mg{\cdot}L^{-1}$		27
Na^+	$mg{\cdot}L^{-1}$		119
Sodium adsorption ratio (SAR)	–		4.3

[a] Longhurst *et al*. (2000)
[b] Wrigley (1994)

The N concentration may vary seasonally. Wang *et al.* (2004) suggest that N concentration reaches a peak within 1 month of the start of lactation and then gradually declines.

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1.3 Yard washdown

Yards are cleaned by floodwash, hand-held hose or high-pressure hydrant. The resulting flow rates affect effluent collection and must therefore be taken into account.

Fixed high-pressure systems

Fixed high-pressure outlets for yard washing can be located at points around the perimeter of the yard, hoses can be installed on backing gates or booms, and hydrants can be installed at select locations around a yard. All rely on the need to convert pressure to kinetic energy, thereby providing an adequate velocity for the entrainment of manure. With fixed high-pressure outlets, the volume of water used and the outlet pressure will need to be much greater than that for hand-held systems unless the fixed system allows for sequential use. Obviously in the case of remotely operated, fixed high-pressure outlets, the demand for labour and time is lower. Multiple-hydrant high-pressure hose systems minimise the length of hose needed to command a milking shed yard. These multiple-outlet connections are convenient, but wear can lead to leakage of water and loss of pressure. Single proprietary hydrant systems which are not reliant on a high-pressure hose are available, applying the washdown water as a surge or fan. They can be installed as single hydrants or as multiple units in larger yards, particularly when the yard exceeds 10 m × 15 m. A range of hydrants and outlets and associated configurations are available, some being available off-the-shelf, others being made by farmers keen to save time.

Pressures

Pressures up to 900 kPa can be used as long as the pressure class of the pipes and associated hose is acceptable. Unless the installed system allows for sequential cleaning, the volume of water used and the outlet pressure will need to be much greater than for hand-held systems. Given the pressures used, the rating number of the pipe needs to be at least PN9 (Class 9), preferably higher. If there is any prospect of water hammer, operate valves in a sequence and avoid butterfly valves unless pressures have been checked.

Specifications and coverage

A 75-mm-diameter pipe operating at a design pressure of 200 kPa could deliver a flow rate of 10 $L \cdot s^{-1}$ to an outlet. At this flow rate, 1 m^2 of yard would be covered by 10 mm of water in a second (but more likely 4 m^2 by 2.5 mm). Under these conditions, the flow velocity through a 25-mm nozzle would be about 4 $m \cdot s^{-1}$. It would be possible to water a yard 10 m × 20 m in a few minutes with good coverage. In 5 min of application at 10 $L \cdot s^{-1}$, the volume applied would be 3000 L. This is approximately 50% greater than what would be achieved with a hand-held hose, which would take about 20 min. The spacing of outlets depends on the size of the yard and the water supply available. Multiple outlets are a feature of large yards, and are typically spaced 5 m apart.

Water quality

Water quality can influence the performance of high-pressure hose systems. Standard practice is to source water with low clogging potential. To minimise particulate matter, install a strainer on the pump suction and, if necessary, as an in-line filter. Centrifugal and helical screw pumps are used for high-pressure hoses (Garrett *et al.* 1991). Both types are prone to damage when pumping particulate matter, particularly abrasive solids such as sand. If the water is saline there is a risk of damage to metallic fittings;

corrosion can be aggravated by the use of recycled water, which tends to have higher salinity than fresh water.

Recycled water can be used for high-pressure washdown if it is drawn from ponds so as to avoid solids and the effluent is treated to a high enough standard. However, the high-pressure application of recycled water can increase the risk of microbial pathogen contact with humans and stock by splashing, and there is an increased risk of aerosols. Floodwashing of surfaces gives a lower risk of spreading aerosols than a nozzle, jet or sprinkler application. The risk of viable microbial pathogens being present in washdown is reduced markedly with increased standards of treatment.

Yard surfaces

The jet of water striking the yard surface can gradually remove fines and dislodge the concrete, a process further aggravated by hoof traffic. The concrete surface receiving the jet must therefore be prepared to withstand scour. Ideally the outlet or hydrant should be designed or periodically moved or redirected to avoid excessive intense localised application. Standard practice is to 'scabble' the concrete or impress the surface with a grid to improve hoof traction, but the rougher the surface is, the more difficult it is to clean. Bunding is essential to ensure that splash from the yard does not erode surrounding earth and to direct the washdown water to a sump. The sump must be sized for the flow without reliance on an effluent pump to control the level of effluent.

Hand-held hoses

Flexible hoses are available in a range of diameters and materials that are resistant to abrasion and damage from exposure to sunlight. Vanderholm (1984) made the following recommendations for hand-held washdown hoses:

- Wash-down equipment should be designed for a flow of 3 to 4 $L{\cdot}s^{-1}$ with 100 to 140 kPa of pressure at the nozzle. Many of the centrifugal pumps now available deliver this quantity of water at the required pressure.
- Place the pump as close as possible to the storage tank to minimise suction lift.
- Calculate the interaction between pipe size and head loss and incorporate it into the design.
- Use a delivery pipe between the pump and the wash-down hose with a minimum diameter of 38 mm.
- Use a wash-down hose with a minimum diameter of 38 mm and a maximum length of 9 m. Provide a delivery pipe with multiple draw-off points to achieve this, if necessary.
- Fit a quick-action valve at each draw-off point and between the hose and nozzle.
- Install an overhead gantry or hooks along the yard wall to lift the hose off the ground during use and for storage.

Pre-wetting yards before milking is recommended to assist cleaning. Supplementary feeding can change the characteristics of manure, and rough concrete yards can impede cleaning. Calculations and recommendations that cover these issues more thoroughly are given in a more recent document from New Zealand Dairying and the Environment Committee (2006).

If the hose is wider than about 75 mm, it can be difficult to lift and manoeuvre, so this is the usual size limit. Hose sizes range commonly from 38 to 50 mm. The combination of hose diameter, orifice size and available pressure dictates flow rate. Operating pressures of at least 100 kPa are recommended, and maximum pressures of about 200 kPa are considered safe. Although operating pressures of 50 m (500 kPa) are possible, control of the hose will be difficult, and the hose will need to be firmly restrained when

fitted with a nozzle. The nozzle dictates the velocity of the water jetting out. Once this velocity exceeds about 4 $m \cdot s^{-1}$, manual control of the hose will be difficult, and the jetting water can injure stock, exposed skin and concrete. To minimise these problems, the orifice should not be less than one-third of the pipe diameter. To assist flow control and to avoid water wastage, install a control valve at the end of the hose. Commercial nozzles are available with an integrated valve, which provides an adjustable high-flow or jetting capability.

Where recycled effluent water is used for washdown, the hose should be periodically flushed with fresh water and specially marked for the purpose. If the recycled water or washdown water has a salinity level exceeding 1000 $mg \cdot L^{-1}$ TDS, use stainless steel fittings or high-impact-resistant HDPE fittings. The longer the pipe, the greater the loss of pressure due to friction, and couplers and connections cause additional losses: each connection has an equivalent pressure loss of about 1.5 m of hose.

The amount of water and time needed for cleaning a yard with a hand-held hose depends on:

- surface area
- the roughness of the yard surface
- the slope of the yard surface
- the dryness of the yard surface
- the amount of built-up manure and track material
- prevailing weather conditions
- the farmer’s preference.

A range of studies indicate that there is marked variation in the cleaning volume required for a unit area of yard: between 10 and 40 $L \cdot m^{-2}$; a design volume of 15 $L \cdot m^{-2}$ is common. If we assume that a 50-mm-diameter hose is connected to a hydrant and carries 4 $L \cdot s^{-1}$ at 175 kPa, the hose could be used to clean an area of about 10 m^2 in 5 min. An advantage of high-pressure hose and hydrant systems is the ease of cleaning irregular-shaped yards when floodwashing is unviable; a typical application is a circular yard which can be readily cleaned by four hydrants installed at 90° positions on a ring main around the perimeter. These systems can also be installed on fluted yards where the stock entrance narrows; in such systems the spacing of hydrants increases as the yard funnels outwards.

Safety considerations

High-pressure pipes, bayonet fittings and nozzle connections must be maintained to avoid bursts which could injure workers or livestock. The frequent starting and stopping of water increases fatigue of components, so regular inspection and replacement of worn components is critical to the maintenance of performance. If recycled water is used for cleaning, people should remain clear of outlets to reduce the risk of ingesting aerosols. Periodic flushing with fresh water will reduce the build-up of slime on nozzles and outlets. This cannot be done when workers or stock are in the path of the jetting water, which can travel further than 10 m at 5 $m \cdot s^{-1}$. Although common sense should prevail, accidents can happen, particularly when nozzles are left open as pumps start up.

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1.3 Yard washdown

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1.4 Floodwash systems

Floodwash or flush systems offer large savings in the labour required to clean yards, feedpads and flush alleys (cow movement lanes in freestall sheds). Adopting suitable design criteria is essential to ensuring a thorough cleaning action and avoiding unnecessary water use.

Design criteria

Following a technical assistance program on the NSW Mid-North Coast, Bullock (2002) reported typical yard cleaning rates of 7 to 10 $m^2{\cdot}s^{-1}$ for floodwash systems, compared to 0.3 to 0.4 $m^2{\cdot}s^{-1}$ for manual hosing and 2 $m^2{\cdot}s^{-1}$ for hydrant systems. Therefore, as a labour-saving option, floodwash systems are very attractive and have been adopted widely in new dairy developments.

However, for a floodwash system to effectively clean the surface without scraping, the design criteria selected must reflect the nature of the material to be removed: there is a considerable difference in the flush velocity required to dislodge and transport organic matter compared with sand and stones, or freshly excreted manure compared with dry manure. Wedel (2000) reports that a flush velocity over 0.3 $m{\cdot}s^{-1}$ is sufficient to scour most organic material, but that 1.5 $m{\cdot}s^{-1}$ is needed to move sand particles of up to 5 mm diameter.

Where a floodwash system is being considered for cleaning a feedpad, bear in mind that dry manure adheres strongly to any concrete surface, and that it is impractical to expect that water alone will dislodge it. Mechanical scraping will be required to assist floodwashing in such situations.

Target depth and velocity

The design depth of flow ranges from 25 to 100 mm. Depths of 75 to 100 mm are generally used in freestall flush alleys where manure build-up is heavy. Depths of 25 to 50 mm are typically used for holding yards in Australia.

Table 1. Minimum floodwash criteria.

Situation	Minimum depth (mm)	Minimum velocity ($m{\cdot}s^{-1}$)
Some mechanical assistance may be necessary	25	1.0
Recommended for most yards	50	1.0
Sand-laden manure and freestalls	75	1.5

Manning's equation (1) is commonly used to check that the required flush velocity will be achieved at the target flow depth. It is apparent that yard slope and surface roughness are important determinants of the flush velocity.

$$v = \frac{R^{0.67}\ S^{0.5}}{n} \qquad (1)$$

where v = velocity ($m{\cdot}s^{-1}$)

R = hydraulic radius (m) (see Equation 2)

S = slope ($m{\cdot}m^{-1}$)

n = roughness coefficient.

Values of Manning's *n* range typically from 0.015 to 0.02 for concrete yards and channels: the rougher the surface, the larger the number.

For rectangular channels and yards, the hydraulic radius is calculated by:

$$R = \frac{WD}{W + 2D} \tag{2}$$

where *W* = width (m)

D = depth of flow (m).

Flow rate required to achieve target depth and velocity

Once the flush depth and velocity are known, the flow rate required to generate those characteristics can be simply determined by Equation 3:

$$Q = A\,v \tag{3}$$

where *Q* = flow rate ($m^3 \cdot s^{-1}$)

A = area (m^2)

v = velocity ($m \cdot s^{-1}$).

Figure 1 shows the flow rate required to flush a 12-m-wide yard with a minimum flush velocity of 1 $m \cdot s^{-1}$ and a target depth of 50 mm. A yard slope of 2% (or 1 in 50) achieves the minimum flow rate required (650 $L \cdot s^{-1}$); flatter slopes will require a larger flush depth to achieve the 1 $m \cdot s^{-1}$ flush velocity; steeper slopes will produce higher flush velocities. Both increase the required flow rate.

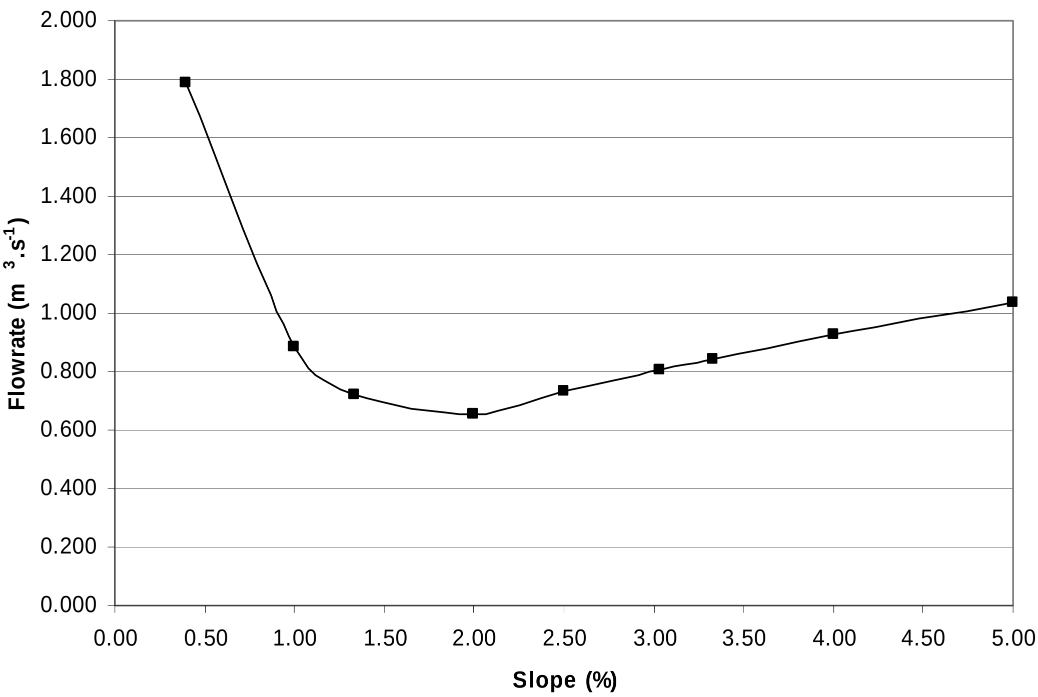

Figure 1. Flow rate required to flush a 12-m-wide yard (*v* = 1.0 $m \cdot s^{-1}$, *D* = 50 mm, *n* = 0.0175).

Tables of required flow rates can be produced for a range of yard slopes and widths. Table 2 shows an example for a flush velocity of 1.0 $m \cdot s^{-1}$ and depth of 50 mm (with Manning's *n* = 0.0175).

Table 2. Flow rate (Q, $m^3 \cdot s^{-1}$) for various yard widths and slopes (v = 1.0 $m \cdot s^{-1}$, D = 50 mm, n = 0.0175).

	Width (m)								
Slope (%)	4	6	8	10	12	14	16	18	20
1.00	0.304	0.450	0.602	0.738	0.887	1.036	1.185	1.334	1.483
1.33	0.245	0.359	0.481	0.602	0.723	0.833	0.939	1.057	1.175
2.00	0.216	0.325	0.435	0.545	0.654	0.764	0.874	0.983	1.093
2.50	0.241	0.364	0.486	0.609	0.732	0.854	0.977	1.100	1.222
3.00	0.264	0.399	0.533	0.668	0.802	0.936	1.071	1.205	1.339
3.33	0.279	0.420	0.562	0.703	0.845	0.986	1.128	1.270	1.411
4.00	0.305	0.460	0.615	0.770	0.926	1.081	1.236	1.391	1.546
5.00	0.341	0.515	0.688	0.861	1.035	1.208	1.382	1.555	1.728

Use ‘volume per metre width’ as a check, not a design criterion

The volume of water required for floodwashing is typically reported as 500 to 1500 L per metre width of yard in Australian references and is based on easily measurable parameters (i.e. water use and yard width) from successful systems. Previously, these recommendations have been used to determine the required flush duration. Instead, it is recommended that designers adopt US procedures for determining the flush duration. Fulhage and Pfost (1993) state that the flush duration must either:

- achieve a minimum contact time of 10 s (suitable for yards and short alleys); or
- maintain the flow rate for sufficient time for the wave front to traverse at least one-third of the alley length (suitable for freestall alleys).

At 1 $m \cdot s^{-1}$, the criterion adopted changes from contact time to the one-third travel time if the yard length exceeds 30 m. That is, the flush duration will be at least 10 s for yards up to 30 m, and the length divided by three times the velocity after 30 m. For a flush velocity of 1.5 $m \cdot s^{-1}$, the critical yard length is 45 m.

Cleaning the 12-m-wide yard examined in Figure 1 would require a minimum flush duration of 10 s if the yard is 30 m long and a volume of 6500 L. Equivalent to a volume of 540 L per metre of width, this falls within the range of reported volumes. It is important to remember, however, that 540 $L \cdot m^{-1}$ is the result of applying the depth–velocity–duration criteria and that, although the volume–width rule of thumb can be used as a check, it is not the starting point for design.

Floodwash system configurations

Once the required flow rate has been determined, the delivery system, including tank, head and pipe configuration, can be designed. The range of floodwash systems includes both pre-fabricated and custom-made installations, but all fall into one of two categories: ‘above the surface’ delivery and ‘buried main and riser’. Large-volume irrigation pumps (axial or mixed flow) are an alternative to using a tank to supply the required flow rate for either delivery configuration.

Above-the-surface delivery

A tank with one or more outlet valves and short delivery pipes is located beside the upper end of the yard. The outlets are oriented to spread water across the yard width or are fitted with a manual direction-control vane. Tanks should be mounted at least 2 m above the yard elevation. Specialist floodwash tanks and used petrochemical storage tanks, which may range from 4 to 8 m in height, can be mounted on a slab at yard level.

Buried main and risers

A main delivery pipe, usually HDPE of 300 mm diameter or greater, with approx. 150 mm risers on 2- to-3 m centres, is fabricated on site and bedded into place before the yard is concreted. The bedding is critical to prevent movement during use, and all junctions require thrust blocks (see Water Service Association of Australia (2002) or similar for thrust block details). Grates or hinged lids must be fitted onto risers to prevent cows stepping into open risers. Tanks may be mounted on a stand or slab to achieve sufficient head.

Buried main and pipe systems are more expensive than above-the-surface delivery systems, but they offer the advantage of uniform delivery of water and cleaning. Additionally, more risers may be located in areas of higher activity, where extra flushing is required. They are suited to wide yards or where the location of the tanks and pipes for an above-the-surface delivery system is restricted; that is, in freestall alleys.

Figure 1. Floodwash tanks and above-the-surface delivery for feedpad flushing.

Where less than the full volume of the tank is used for each flush, a pressure relief or vacuum release valve may be required on the delivery pipe to prevent damage upon

valve closure. Gear-operated, air-actuated or electric valves are preferred over lever-type valves, as they can prevent the valve from being opened or closed too quickly and causing water hammer. In the case of air-actuated valves, two-way actuators provide the most flexibility, as they can be set up to control the speed of opening and closing, unlike one-way actuators, which have spring-operated return-to-close valves.

Hydraulic design procedure

Tools are available to help designers size pipes and valves for floodwash delivery. Skerman (2004) provides a spreadsheet that enables such calculations to be performed, including the response of flow rate to decreasing available head.

If a manual calculation is necessary, the procedure can be summarised as follows:

- Select appropriate flow criteria (depth and velocity) from Table 2.
- Using the target flow depth, calculate the flush velocity using Equations 1 and 2.
- If the velocity is less than required, increase the depth and recalculate until the target velocity is achieved.
- Calculate the flow rate using Equation 3.
- Determine the tank and discharge pipe and valve configuration (or pump–pipe arrangement for direct pumping) and select appropriate friction and local losses.
- Solve the continuity equation (see standard hydraulic texts) for pipeline velocity (by trial and error) according to the head available from the tank.
- If the target flow rate cannot be achieved for the head available, select a larger pipe diameter or alternative (higher or larger) tank arrangement.
- Calculate the flush duration to deliver the larger volume determined by adopting a contact time of 10 s or maintaining the target flow depth and velocity over at least one-third of the alley length.
- Determine the flush volume and confirm that the tank provides sufficient head for the flush duration.

Floodwash systems and the effluent system

Adopting a floodwash system will have implications for the remainder of the effluent management system. There are several implications:

- A larger volume of water is required than for any other yard cleaning system. McDonald (2005) reports that the average volume for floodwash tanks in Victoria in a 2005 audit was 17 000 L, and that most farms used the full capacity in flushing. The reported volume used in floodwashing yards ranged from 8000 to 60 000 $L{\cdot}day^{-1}$. Fortunately, floodwashing systems can use treated effluent drawn from the terminal pond in a multiple pond system to minimise the use of clean water. However, solids separation traps and pumps will have to be sized larger than otherwise required to cope with the increased throughput.
- A solids separation trap or large-capacity gutter should be used to collect the flush at the lower end of the yard. Solids separation traps must be sized to accommodate the volume of the floodwash tank in addition to the volume of sludge built up between cleaning events (see chapter 2.1 'Solid–liquid separation systems'). Gutters conveying floodwash away from yards to traps or ponds must be designed to convey the maximum flow rate without backing water onto the yard.
- There is potential for salt build-up in systems using recycled effluent. A reduction in clean water input to the effluent system, in conjunction with the additional pond surface and evaporation from the multiple pond systems required, will increase

the salinity of the effluent, potentially compromising anaerobic pond function and exacerbating salinity problems following land application (see chapter 2.3 ‘Anaerobic, aerobic and facultative ponds’).

- Struvite (a build-up of the salt magnesium ammonium phosphate) may become a problem where the ratio of recycled effluent to clean water entering the effluent system is high. The presence of struvite can be an indication that TDS levels are too high and that additional clean water needs to be added to the system. See Hopkins (2002) for further information on struvite and its control.
- Some farms draw effluent from sumps or solids traps to supply the floodwash, rather than from the terminal pond in a treatment system. Although anecdotal evidence suggests that this reduces odours, this approach also causes algal growth and slippery yard surfaces.

There are implications for yard design as well:

- Nib walls or kerbs must be a minimum of 200 mm high to contain the wave produced.
- The flush should discharge from the end of the yard into the gutter or solids separation trap. However, if there is no alternative to a side discharge arrangement, yards may have a cross fall (<25 mm at the upper end, increasing at the lower end), but be aware that the high side will not clean well. Cross-drains are an alternative but may allow some solids to drop out of suspension where the flush slows to change direction.
- Valves and outlet rudders for above-the-surface delivery must be guarded to prevent injury to cows on the yard. Outlets must be positioned so that they do not reduce good cow flow.

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1.5 Sump design

Sumps are located within or at the end of the paved surface of dairy yards or feedpads to collect washdown water and rainfall runoff for gravity discharge or for pumping. The collected effluent is then diverted to a land application area, to storage or to a treatment facility. These sumps can be small to ensure entrainment of solids or large to serve as a pre-treatment facility and even emergency storage. As sumps are generally in-ground and can be confined spaces, they need to be designed for secure and safe access.

The rational design of sumps for collecting dairy washdown effluent is limited by the variability of effluent characteristics: although volumes are predictable, the amount and type of solids contained in them varies seasonally and diurnally, warranting flexibility in sump design. The principal objectives need to be either:

- the entrainment of solids through maintenance of agitation and elevated velocity by using a relatively small chamber and sometimes a mechanical agitator with rapid pumped removal of effluent, or
- the separation of solids through deposition in a chamber or series of interconnected chambers, or filtration via grilles with periodic dewatering by pumps or gravity release (see chapter 2.1 'Solid–liquid separation systems').

Stormwater diversion is dealt with in chapters 2.2 'Direct application systems' and 2.6 'Effluent storage requirement'.

There are few hard and fast rules for sump configuration, but experience indicates that they need to be structurally sound, able to withstand impact and make use of gravity wherever possible. Although large-diameter pipes or even circular water troughs can be placed vertically as sumps, this practice does not encourage ease of cleaning or maintenance, and does create confined entry conditions, requiring special OH&S consideration.

Large sumps should have:

- an overflow to divert effluent to a bunded area
- no sharp corners or dead spots
- stone traps and debris grilles which can be removed
- stormwater diversion
- sloping or conical floors to assist desilting, desludging and emptying
- adequate mass or restraint in high-water-table environments
- a secure cover or fence which can be unlocked for access but cannot trap personnel
- warning signs if confined entry conditions dictate
- agitation if retention of effluent exceeds 30 min
- footholds or a ladder for access, particularly if smoothed-walled
- ramped access to facilitate the removal of solids with a front-end loader.

As there are a variety of dairy yards with different inlet conditions, effluent characteristics and approach velocities, there need to be a range of sump configurations. Some sumps work successfully on one farm but not on another, and slight changes in dimensions have often been counterproductive. Field observations show that you should not rely on grilles and weeping walls in isolation for solids removal. If a pump is used, provide adequate storage for effluent in the event of a power failure. The duration of an outage depends on the local power grid, but 48 h is possible in some rural areas, and 24 h is not uncommon.

Sump configuration

Small sumps collecting effluent without an agitator should be designed with sloping floors (45°) to direct settled solids directly to the pump inlet. The depth of the sump should allow the pump casing to either readily rest on the bottom when there are supports, or to be suspended 30 to 50 mm above the sump floor.

According to Vanderholm (1984), larger sumps can range in shape from long rectangular to deep cylindrical tanks; for efficient mixing, cylindrical tanks are better. The liquid depth should be about half the diameter or length of the sump. Avoid cylindrical sumps of small diameter with snug-fitting pumps.

Useful guidelines for determining the configuration of sumps and pump intakes are provided in APMA (2001); these guidelines aim at maintaining the life of pumps, but assume minimal solids in the effluent to be discharged.

Pump manufacturers recommend a maximum number of pump starts per hour. Therefore, the minimum sump volume is set by the flow rate into the sump. A one-duty fixed-speed pump starts most frequently when the flow rate is half of the pumping rate, where the cycle time equals:

$$240 \times \text{sump volume (L)} / \text{pumping rate } (\text{L}\cdot\text{s}^{-1})$$

Float switches are commonly used to operate pumps servicing a sump. Load cells are used as an alternative on newer installations.

If direct application is allowed (see chapter 2.2 ‘Direct application systems’), the sump can be designed to impound effluent for up to a week to cater for wet weather, pump malfunction or power outages. Under these conditions, odour is likely.

Sumps and pumps

The minimum depth of submergence of a pump inlet in a circular sump should be 1.5 × D (diameter of the suction line), and the inlet should be located off-centre, 0.25 × D from the sump wall and 0.5 × D from the floor level. Further rules for the hydraulic design of sumps are provided in various texts (APMA 2001, Dicmas 1987, Karassik *et al.* 1976, Sanks 1989, Stepanoff 1976, Yedidiah 1980). The geometry of a sump is important for conveying effluent and avoiding vortex action and the deposition of solids. The sump must be large enough to meet the requirement for the number of pump starts per hour but not too large that solids can settle out (unless agitation is provided).

Waterborne debris and pumps are incompatible. Solids such as gravel, fencing wire, baling twine, sticks and horns can destroy pumps. To ensure the longevity of an effluent pumping system, it is essential to screen out debris or remove it via a grille, trafficable sump or settling chamber before the effluent enters a pumped sump.

Many vertically and horizontally mounted effluent pumps are mounted on beams above or next to the sump for stability and to provide ideal inlet flow conditions. These beams must be rigid and fixed in place to maintain the pump in the correct plane of operation. Avoid cantilevered beams for pump support unless they are very rigid and preferably propped. A poor design will very likely:

- restrict the performance of the pump
- contribute to, or cause, severe damage to the pump, transmission system, electric motor or switchgear
- shorten the useful working life of the pump.

Adoption of the following guidelines should ensure a successful pump sump:

- Avoid turbulence of cascading flows from the inlet pipeline, channel or stream to minimise air entrainment, as this can reduce pump and delivery line performance.

1.5 Sump design

If a long fall cannot be avoided, there must be sufficient length and depth to the pump inlet to compensate.

- Where inlet pipes are <0.5 m in diameter, which is usually the case unless a floodwashing system is used, the installation of a down-turned vertical suction line is good practice.
- The entrance velocity at the mouth of the inlet should not exceed 1 $m \cdot s^{-1}$, and the maximum velocity of water in the suction pipe should not exceed 1.5 $m \cdot s^{-1}$.
- The approach velocity of effluent in the sump should be <0.3 $m \cdot s^{-1}$.
- The pump inlet needs to be adequately submerged to avoid vortex formation.
- Avoid locating the pump inlet in the middle of a small circular sump with a high entry flow rate.
- Avoid sump designs that promote rotational movement of effluent, the possibility of vortex formation or the entrainment of air.
- Keep the inlet submerged to reduce friction losses, maintain submergence, and reduce the risk of cavitation (the rapid formation and collapse of vapour pockets in regions of low pressure—a frequent cause of serious structural damage to pumps).

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1.6 Pipes

Effluent can be conveyed by channels or pipes under gravity or in dedicated pipelines under pressure. If pipes are used, pressure rating, water-hammer and excessive friction losses must be considered during the design stage, and the deposition of entrained materials must be avoided. *Plumatella repens* (a bryozoan animal) and algae are biological agents that can build up on pipe walls and impede flow, and other organisms can attack pipe material, particularly concrete. Struvite (magnesium ammonium phosphate) is a crystalline chemical compound which can also constrict pipelines.

Gravity conveyance in pipes

The choice of pipes, often with grated entries, as opposed to surface channels or drains, to carry effluent needs to be considered in the light of each individual site and situation. The propensity for blockages in gravity pipes needs to be considered, along with the associated restricted access and OH&S issues. Reinforced concrete, PVC (polyvinylchloride) or HDPE (high-density polyethylene) pipes are commonly used for gravity conveyance, and a host of recycled plastic products are also used. PVC is more common, but HDPE pipes can be welded on site and are resilient and easily installed. Concrete and recycled plastic pipelines, though less common, depend on price and contractor preference.

The principal consideration for gravity conveyance is the presence of solids in the liquid. Tables 1 and 2 give recommended grade requirements of pipelines with and without solids in the effluent stream. Ideally, solids should be removed before pipeline conveyance. At least a threefold increase in velocity is required for satisfactory gravity discharge in a pipeline conveying solids relative to effluent without solids. The minimum pipe gradient for conveying effluent with solids therefore needs to be increased at least fifteen-fold relative to solids-free effluent. The minimum recommended pipe diameter for gravity conveyance of raw effluent on a farm is 100 mm based on operating experience. This is dictated not by hydraulics, but by the propensity for blockages. Sewer-class pipes should be used rather than stormwater-class pipes given the need for thicker walls and more resilience under impact.

Table 1. Minimum grades for gravity pipe drains conveying treated pond effluent.

Inside diameter (mm)	Minimum grade (%)	Velocity at full flow ($m \cdot s^{-1}$)
75	0.2	0.29
100	0.1	0.25
125	0.07	0.24
150	0.05	0.23

Table 2. Minimum grades for gravity pipe drains conveying raw effluent from yards or sumps.

Inside diameter (mm)	Minimum grade (%)	Velocity at full flow ($m \cdot s^{-1}$)
75	3.3	1.0
100	2.5	1.0
125	2.0	1.0
150	1.7	1.0

The data in Tables 1 and 2 are based on municipal effluent practice. Their successful application to dairy effluent has been demonstrated over many years. Wedel (2000) has, however, determined more appropriate critical velocities for control of scour and deposition of sand-laden manure, confirming the need for transmission velocities

exceeding 1 m·s^{-1} to avoid deposition. Obviously, particle size is the major control, and the faster the passage of effluent, the larger the particle conveyed. To avoid the transport of sand, gravity removal is essential before diversion to a sump or storage.

Blockages are common in gravity drainage pipes, and the abovementioned gradients are minimum values only. The effectiveness of pipes for conveyance of effluent is dictated by the effectiveness of grates and mesh or sedimentation sumps for removing entrained solids.

Inspection pits should be installed at discrete intervals in any gravity drainage pipeline. Recommended spacings vary with diameter, but 20 m is common. These pits allow for venting and access for cleaning. Common practice is to use pits at any change of grade or direction, although T-junctions can be used to facilitate access as well.

Pressurised conveyance

Pipes are manufactured in a range of pressure classes, expressed in terms of a PN rating (nominal pressure rating at 20 °C). Table 3 details the common pipe classes for rural applications. The choice of design pressure needs to consider the impact of increased temperature, which will reduce the pressure rating for PVC and HDPE pipes. In addition, an allowance must be made for water-hammer. In the absence of a detailed investigation, it is prudent to opt for an increase in pressure class to allow for high-risk situations where high-pressure pumps stop and start without surge protection.

Example

PN 4.5 (Class 4.5) pipe has a pressure rating of 450 kPa. Water-hammer (without surge protection) must be allowed for when you are selecting pipes which will be subject to rapid opening and closing of valves or starting and stopping of pumps. A hydraulic system requiring 400 kPa to be carried by the pipeline would require a PN 6 pipe—a pipe with a pressure rating of 600 kPa—to allow for water-hammer. This is based on experience and conservative practice.

Table 3. Pipe classes.

Imperial	Metric
Class A—150 ft head = 65 psi	PN 4.5 (Class 4.5) = 4.5 atmospheres = 450 kPa = 45 m head
Class B—200 ft head = 87 psi	PN 6 (Class 6) = 6 atmospheres = 600 kPa = 60 m head
Class C—300 ft head = 135 psi	PN 9 (Class 9) = 9 atmospheres = 900 kPa = 90 m head
Class D—400 ft head = 175 psi	PN 12 (Class 12) = 12 atmospheres = 1200 kPa = 120 m head

Table 3 is provided to assist in conversion of Imperial to metric units. It is still common to find pumps, pipes and pressure gauges marked in Imperial units, because the US irrigation market is dominant, and many contractors and farmers retain the old units. Some contractors have a personal preference for particular types of pipe, and as long as they have no pecuniary interest, their preferences should be catered for.

If small-diameter pipelines are to be used, it is wise to be able to flush the line with water. The life expectancy of PVC and HDPE pressure pipes is not really dictated by the corrosive nature of effluent, but reinforced concrete pipes react to sulphuric acid. Over time detritus builds up on pipe walls, causing friction losses, and sometimes poor bedding conditions contribute to abrasion of external walls. Poor bedding can also lead to pipe collapse and leaking at joints.

Types of pipe and fittings

Table 4 compares the properties of HDPE and PVC pipes. Although not commonly used for effluent conveyance, many other types of pressure pipe are available, including:

- galvanised iron
- cast iron
- ductile iron
- recycled plastic
- aluminium
- concrete.

Effluent can be corrosive when in contact with metallic or concrete pipes. If effluent contains sulphurous compounds from soil or cattle feed, corrosion of concrete can be accelerated. Concrete with a structural strength of 30 MPa should be used for pipes and hydraulic structures. The minimum reinforcement cover should not be less than 30 mm. Sulphate-resistant cement is recommended, and all exposed starter bars must be galvanised. Steel, cast iron and ductile iron pipes should be protected from corrosion.

Pipe fittings are commonly made in the same material as pipes. The mixing of different types of pipe and fittings is not recommended, owing to different rates of expansion and contraction and dimensional intolerances. Energy losses in fittings are due to frictional resistance, and these must be accounted for in any hydraulic analysis. The fittings not only dissipate energy, but also enable the build-up of debris, which will throttle flow. Ideally, any fitting or pipe join should avoid a sharp edge. The use of external fittings avoids internal obstructions.

Table 4. Comparison of the properties of pipes.

Property	Polyethylene (HDPE)	Polyvinylchloride (PVC)
Flow characteristics	Good	Good
Weight	Light	Light
Available lengths	Up to 300 m	6 m
Corrosion	No	No
Flexibility	Yes	Partially
Effect of temperature on strength	Sensitive	Sensitive
Installation	Simple	Simple
Impact resistance	Fair	Fair
Sunlight	Can withstand	Requires protection
Life expectancy	50 years	50 years
Cost equivalent	1 unit	1.5 to 2 units

Pipe cover, alignment and protection

Pipelines should not be located above ground where human, vehicle or animal traffic is encountered. Provide at least 300 mm cover for pipes in trenches. The larger the pipe and greater the traffic load, the greater the cover required.

The following are recommended minimum installation depths for Australian farming practice, but it is wise to ensure that Australian Standards are adhered to for specific types and pressure classes of pipe:

- open grazing country 300 mm
- garden 300 mm
- roads 400 mm
- cultivated ground 500 mm.

1.6 Pipes

Take care with the transportation, handling and installation of pipes, and follow manufacturers' laying procedures, particularly bedding specifications. Much greater care is required with the installation of gravity pipelines than with pressure lines, as these need to be installed at grade and cannot afford to settle. Manufacturers frequently recommend more conservative depths; commonly PVC is installed with 600 mm cover under sealed roads and 750 mm under unsealed roads. Under major roads carrying B-double vehicles and milk tankers, some councils require both PVC and HDPE pipes to be sleeved in steel or concrete.

The relevant Australian Standards are:

- AS/NZS 1254:2002 PVC pipes and fittings for storm and surface water applications (Standards Australia 2002b)
- AS/NZS 1260:2002 PVC-U pipes and fittings for drain, waste and vent application (Standards Australia 2002a)
- AS/NZS 1477:2006 PVC pipes and fittings for pressure applications (Standards Australia 2006b)
- AS/NZS 2032:2006 Installation of PVC pipe systems (Standards Australia 2006a)
- AS/NZS 2033:2008 Installation of polyethylene pipe systems (Standards Australia 2008)
- AS 2439.1–2007 Perforated plastics drainage and effluent pipe and fittings—Perforated drainage pipe and associated fittings (Standards Australia 2007a)
- AS 2439.2–2007 Perforated plastics drainage and effluent pipe and fittings—Perforated effluent pipe and associated fittings for sewerage applications (Standards Australia 2007b)
- AS 2698.2–2000 Plastics pipes and fittings for irrigation and rural applications—Polyethylene rural pipe (Standards Australia 2000)

Installers should seek guidance on pipe size, gradient and laying procedures from suppliers and contractors so that installation costs can be ascertained before the pipe and fittings are purchased. The cost of installation may well overshadow the cost of the pipe. If shallow rock, cracking clay or elevated water tables are present along a pipe route, very high installation costs are likely. Factors to consider with costing the installation of pipes include:

- constraints on availability owing to very high demand in district or region
- lack of competition in supply of pipes or between construction contractors
- quantity of pipes and reduced unit cost with increased scale
- weather during installation
- the deeper the pipe, the greater the excavation cost and the more limited the prospect for ploughing in
- the availability of bedding
- quality control for backfill
- water table height
- the amount of rock to be excavated.

Of paramount importance is the need to consider an appropriate alignment. Before excavating in unfamiliar ground, call the 'Dial Before You Dig' service to determine the type and proximity of services. In some jurisdictions a cultural heritage investigation will also be necessary.

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2.1 Solid–liquid separation systems

Solids separation systems are becoming an increasingly important component of effluent management systems, particularly for larger herds. Removing solids from the effluent stream offers improved system reliability and reduces sludge accumulation in effluent ponds.

Why install a solids separation system?

Solids separation systems offer the following advantages:

- They minimise the need for agitation in sumps and reduce the likelihood of blockages in pumps and pipes.
- They reduce the rate of sludge accumulation in ponds. Together with the reduction in volatile solids (VS) loading to the pond, this allows smaller ponds to be built or extends the life of existing ponds.
- They allow the use of conventional irrigation equipment for distribution of effluent from adequately sized single ponds (although high salinity levels are not reduced, and some equipment, for example centre pivots, may require additional protection).
- They concentrate organic matter (and nutrients to a limited extent) for direct application to pasture, composting or cost-effective transportation off-site.

However, installing a solids separator may also introduce some additional requirements:

- A solids handling system (separator, impermeable storage pad, front-end loader, spreader etc.) will be needed in addition to the existing liquid handling system, introducing additional energy, labour, repair and maintenance costs.
- Separated solids will generally have a total solids (TS) content of 10% to 30%. Effluent may drain from wetter storage piles and, along with any rainfall runoff from the pad, must be collected to drain back into the effluent management system.
- Separated solids will become anaerobic and may emit odours unless composted or dried to a moisture content of <60% (40% TS). The dry crust on stockpiles limits odour emissions until the stockpile is disturbed.

As most of the precursors to odour generation (carbon compounds, proteins and nutrients) are contained in the finer particle fraction, which is not removed by gravity or mechanical systems, solids separation has only a limited capacity to reduce odour generation. The reduction in VS loading may reduce odour generation in an existing pond, but quantitative data is lacking. During the design phase for new ponds, a reduction in VS loading rate may result in a smaller pond surface area which will reduce expected odour emissions. If significant reduction of odours and nutrients is an objective, additional chemical treatment may be necessary (see chapter 5 'Odour emissions and control').

Types of solids separation systems

Solids separation systems can generally be divided into two broad categories:

- those that rely on gravity (trafficable solids traps, sedimentation basins and ponds)
- mechanical systems using screening (inclined stationary screens, elevating stationary screens, vibrating screens, rotating screens), centrifugation (centrifuges, hydrocyclones) or pressing (roller press, belt press, screw press).

Gravity sedimentation systems

Settling or sedimentation of solids by gravity is the most effective method for separating solids from dilute effluent streams such as dairy shed effluent, loafing pad or feedpad runoff, and manure flushed from freestalls). Sedimentation systems can consistently remove more solids and nutrients from effluent than mechanical methods when the TS content is low, and remain the favoured approach for the dilute effluent typical of Australian dairies. Sedimentation systems are not suited to effluents with a TS content exceeding 3% (Mukhtar *et al.* 1999), and settling rates may become hindered when TS > 1% (Sobel 1966).

Sedimentation basins are typically shallow structures designed to achieve a low through-flow velocity and accommodate the accumulated settled material between periodic clean-outs. Trafficable solids traps, now common on many dairy farms, are a form of sedimentation basin using a concrete base for regular clean-out by front-end loader. Earthen sedimentation basins are a more suitable option where the catchment area (holding yards, loafing pads or feedpads) will generate a significant volume of runoff during storms. Sedimentation ponds are deeper structures that do not drain before clean-out.

Trafficable solids traps

Guidelines for the construction and management of trafficable solids traps are provided in existing guides (DairyCatch 2006, Haughton 2006, NSW Dairy Effluent Subcommittee 1999, Skerman 2004) and via the Target 10 website at www.nre.vic.gov.au/cgi-bin/exsysweb.exe?KBNAME=fer03. In summary, the design of trafficable solids traps should include the following provisions:

- The trap must have enough capacity for the solids that accumulate between cleanouts (see Table 4), plus the volume used in yard washing each milking (critical when floodwashing), plus freeboard of 200 mm to avoid spills upon clean-out of wetter-than-normal settled solids.
- The ramp slope must not exceed 10:1 (horizontal to vertical) even at sites where a 4WD tractor is used.
- The trap must have a minimum width of 3 m (or tractor width plus 0.6 m).
- The trap must have a liquid depth of no more than 900 mm.
- The permeable weir must be placed so that the drainage 'path' to discharge is no more than 12 m (Harner *et al.* 2003); large-capacity traps may need the permeable weir to extend along the side of the structure.
- The area of the permeable weir should be as large as the structure allows.
- The spacing between boards (or other permeable weir members) should be adjustable from 10 to 25 mm.
- Where boards are used, orientate them vertically if possible (as straw floats horizontally); see Figure 1.
- Locate the point of entry of effluent into the trap towards the ramp end (as entry near the weir may resuspend settled solids). Heavier material such as sand may be removed first and handled separately if effluent enters at the ramp end.
- Wastewater from the milk room or pit may enter the trap behind the permeable weir (or bypass the trap completely).
- Where the effluent passing through the trap must be pumped out, the sump should be designed so that the pump cut-in level avoids backing water into the settled solids.
- Align the outlet pipe towards the pond to avoid bends.

- Any grooves formed during construction to assist with traction on the ramp must not have raised protrusions that may catch the leading edge of the tractor bucket, and should be oriented at an angle to the slope.
- Fence off solids traps to exclude people, especially children, and stock.
- A drying and storage pad may be required for the solids removed; the pad should drain back into the trap.

Generally, the larger the trafficable solids trap the more effective it will be at removing solids. It is possible to remove 50% of total solids from the effluent stream (see ‘Earthen sedimentation basins and ponds’ below), but to do so, trafficable solids traps should have a capacity of 1.0 m^3 per cow y^{-1} for cows producing 16 L of milk, and 1.2 m^3 for cows producing 36 L (see chapter 1.2 ‘Characteristics of effluent and manure’) and assuming that 10% of manure is collected (Table 4). As existing Australian guidelines range from 0.2 to 0.9 m^3 per cow y^{-1}, either an increase in capacity is needed or farmers must clean traps frequently to be effective. Guidelines from the USA (Fulhage 2003, Harner *et al.* 2003, Midwest Plan Service 1985) suggest designing traps with a volume of 0.9 to 1.6 m^3 per cow y^{-1} if 10% of manure is collected. These values exclude any allowance for freestall bedding, which may contribute another 8 to 11 m^3 per cow y^{-1} where 100% of the waste bedding is collected.

Smaller traps may be appropriate where the objective is to separate only the gravel and sand from the effluent stream. In such case, a ‘weeping weir’ arrangement (where effluent seeps between the horizontal or vertical members of a drainage wall) is not necessary.

Table 4. Solids accumulation rate ($m^3 \cdot week^{-1}$) per 100 cows based on removal of 50% of solids.

Time held (h)	Milk yield ($L \cdot day^{-1}$)						
	16	20	24	28	32	36	40
1.5	1.2	1.3	1.3	1.4	1.4	1.5	1.5
2	1.6	1.7	1.8	1.9	1.9	2.0	2.1
2.4 (10%)	2.0	2.0	2.1	2.2	2.3	2.4	2.5
3	2.5	2.6	2.7	2.8	2.9	3.0	3.1
4	3.3	3.4	3.6	3.7	3.8	4.0	4.1
5	4.1	4.3	4.4	4.6	4.8	5.0	5.2
6	4.9	5.1	5.3	5.6	5.8	6.0	6.2
8	6.5	6.8	7.1	7.4	7.7	8.0	8.3
10	8.2	8.5	8.9	9.3	9.6	10.0	10.3
12	9.8	10.2	10.7	11.1	11.5	12.0	12.4
18	14.7	15.4	16.0	16.7	17.3	17.9	18.6
24	19.6	20.5	21.3	22.2	23.1	23.9	24.8

The table assumes a density of 1000 $kg \cdot m^{-3}$.

For further information on estimating TS, see chapter 1.2 ‘Characteristics of effluent and manure’.

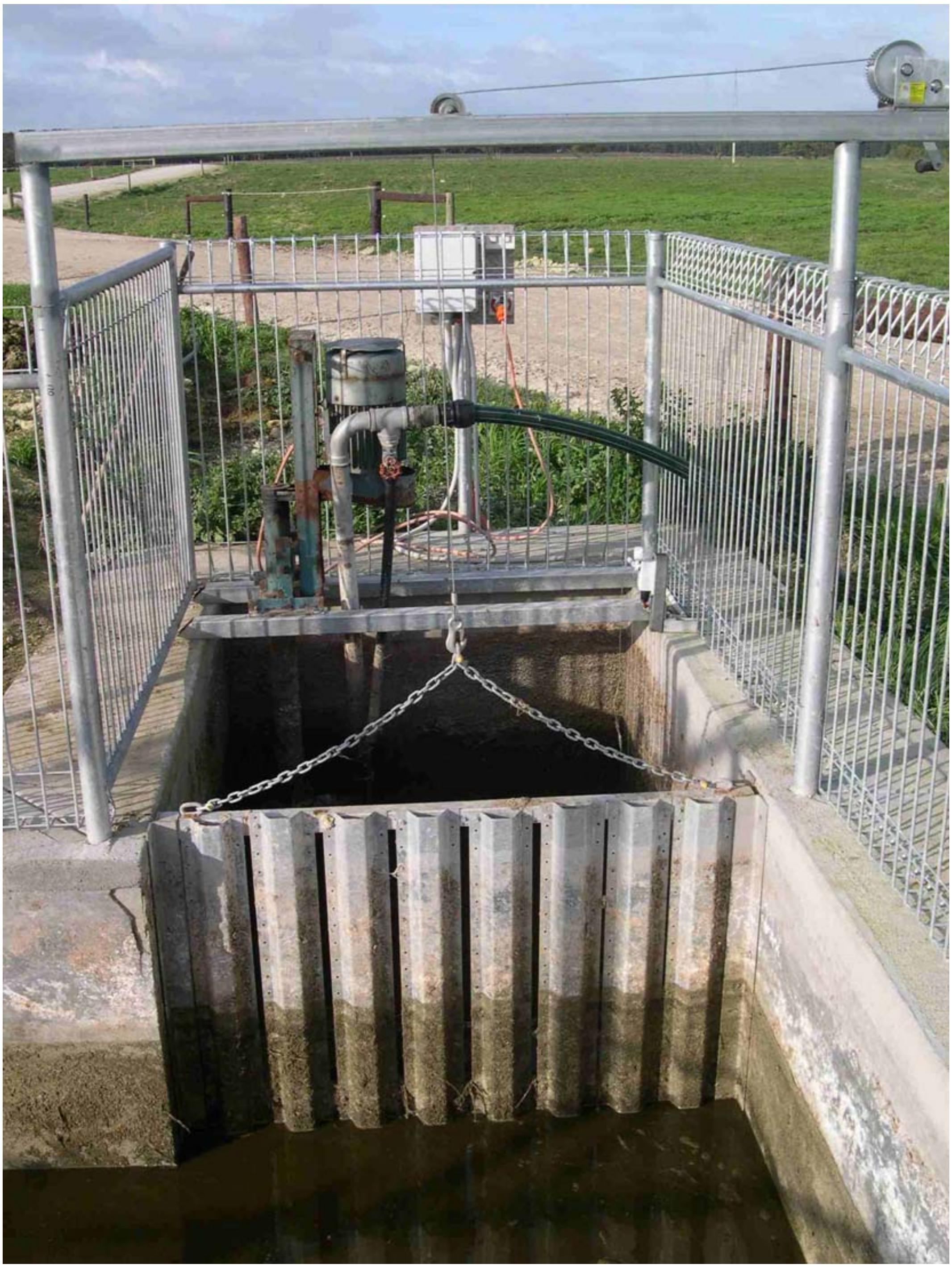

Figure 1. Vertical weir in trafficable solids traps (courtesy of Rural Solutions SA).

Earthen sedimentation basins and ponds

Large, shallow sedimentation basins with an earthen floor are widely used for removing settleable solids from runoff in beef feedlots. The larger treatment volumes afforded by sedimentation basins and ponds also makes them suitable for handling the storm discharge from holding yards, loafing pads and feedpads.

Sedimentation basins are typically designed to drain completely so that the material removed during clean-out can be handled as a solid. A pair (or more) of sedimentation basins with provision to divert flows to one while the other dries provides the opportunity to maximise the solids content of the material removed. Sedimentation

ponds are usually overflow-type structures from which the solids are removed as a slurry with an excavator or specialist slurry pumping equipment following agitation.

The design of sedimentation basins with an earthen floor must take into account equipment requirements for clean-out. As the floor of the structure is unlikely to dry enough to support vehicular traffic, earthen basins should be narrow enough for an excavator to reach to the centre line of the floor, with a width limited to around 15 m for a 1-m-deep basin (and 13 m for a depth of 2 m). Longer-reach excavators are available but are not usually suitable for frequent desludging. It is also important that any compacted liner on the floor and walls of the basin be protected from damage during clean-out. Options include a 150-mm layer of sand or gravel, and recycled tyres placed horizontally across the surface.

Where agitation and pumps are to be used, stable machinery access points must be incorporated into the design (see chapter 6 ‘Occupational health and safety’).

Laboratory research has demonstrated that the majority of settling in static water occurs within 30 to 60 min, typically removing 40% to 60% of TS, 45% to 65% of VS, 30% to 50% of total phosphorus (TP) and 20% to 40% of total Kjeldahl nitrogen (TKN). Settling has little effect on the soluble nutrients ammonium (NH_4^+) and potassium (K) (5%–20%). The solids content of the material recovered typically ranges from 10% to 25% in published results.

Sedimentation and evaporation ponds (SEPs)

Long, narrow sedimentation basins and ponds have been used on dairies in the past and, more recently, have been constructed at larger dairies and freestall developments where large volumes are generated by floodwashing of yards and alleys. Some of these are deep (1.8 m), have no drainage and retain liquid upon clean-out; others use only one or two ponds and are difficult to manage.

Recent research from the pig industry has identified that SEPs offer significant improvements in the recovery of solids removed from effluent (Payne *et al.* 2008). Anecdotal evidence also suggests that SEPs reduce odour when conventional anaerobic ponds are replaced.

The standard SEP system developed by the pig industry comprises three parallel ponds, typically 6 m wide (5–10 m) and 0.8 m deep (0.7–1.0 m). Each pond must be separated by sufficient distance to accommodate an excavator and truck during clean-out (minimum 10 m). Their length depends on the sludge volume and may range up to 600 m. Larger volumes can be achieved by building in multiples of three.

Effluent is directed to one of the three ponds. Once that pond is full of solids, it is taken out of use and allowed to dry out over summer. On account of its narrow, shallow structure, the sludge dries readily and can be removed by an excavator and truck. At any point in time, one of the ponds is filling, one is full and drying, and one is ready to be, or has been, cleaned out.

The equivalent design volume for a dairy cow is approximately 0.6 to 0.9 m^3 per cow (producing 6 kg VS day^{-1}; 10% to 15% collected) for 6 months of operation. Therefore, a 200-cow herd with 15% of excreta collected would require three 30-m-long structures, or approximately 0.6 ha after allowance of 10 m for access between structures. An 800-cow dairy with a feedpad (say, 40% collected) would require three 300-m-long structures.

Payne *et al.* (2008) showed that SEPs give similar results in terms of effluent treatment to conventional anaerobic ponds, with a reducing TS by 79%, VS by 82%, P by 89%, TKN by 36%, and K by 4%. The cost of removing the solids recovered ranged from \$4 to \$6·t^{-1} for large piggeries, compared with \$21 to \$60·t^{-1} for other approaches (sludge pumping, excavating slurry).

Advantages of SEPs:

- Ponds are easily desludged.
- Total odour emissions are reduced compared with conventional anaerobic ponds.
- nutrients contained in sludge are recovered more frequently (6-monthly to yearly).
- Nutrient reuse can be decoupled from treatment where off-site use is necessary.

Disadvantages of SEPs:

- Their shallow depth means a large land area is required.
- The soil must be suitable for the construction of a low-permeability clay liner (or an artificial liner must be used).

The summer drying period is important to produce a low-moisture-content solid, so the use of SEPs may not be appropriate for summer-dominant rainfall areas, and is not appropriate for high-rainfall areas owing to the large surface area. Rainfall–evaporation modelling may be needed for those sites where effluent is to be reused for yard and feedpad washdown.

Designing for settling velocity

The size of sedimentation basins, particularly those receiving storm runoff, is determined by settling characteristics, not detention time. Although some organic solids will start settling as the horizontal velocity of the liquid drops to <0.3 $m{\cdot}s^{-1}$ (as opposed to sand, which will settle below 0.6 $m{\cdot}s^{-1}$), the horizontal velocity should be reduced to less than the suspended solids' vertical settling velocity if maximum removal is to be achieved (the velocity at which sediments are re-entrained is about equal to the their settling velocity).

Settling velocity is influenced by the size, density, shape, and roughness of the particles. Using beef cattle faeces and feedlot manure, Lott and Skerman (1995) established that 60% to 85% of settleable solids will be removed at a horizontal velocity of 6 $mm{\cdot}s^{-1}$ (22 $m{\cdot}h^{-1}$). In the absence of head–discharge curves for permeable weirs, where discharge is restricted by manure, maintaining a flow depth of 100 mm in a 3-m-wide trafficable trap would be sufficient at uniform flow and a discharge of 2 $L{\cdot}s^{-1}$ (50% of a typical washdown hose flow rate). Trafficable solids traps should therefore be cleaned out before the accumulated solids encroach within 100 mm of full.

Where sedimentation basins receive larger flows—such as floodwash or storm runoff from large catchment areas such as loafing pads and feedpads—sizing must account for the expected flow rate. The National Guidelines for Beef Cattle Feedlots in Australia (ARMCANZ 1997) stipulate designing sedimentation basins for a 1-in-20-year design storm using a coefficient of runoff of 0.8 and show an example calculation. Australian Rainfall and Runoff (IEAust 1987) and chapter 2.6 'Effluent storage requirement' of this database outline methods to determine the design storm and resulting runoff volume. An overflow weir or spillway will be necessary to handle events larger than the design storm volume.

Note that the flows from floodwash systems, particularly from the long alleys in freestalls, will be much less at the sedimentation basin inlet than at the floodwash valve. However, little data is available to assist with developing typical design parameters at this stage.

Mechanical separation systems

Inclined stationary screens

Inclined stationary screens have a header tank at the top edge of an inclined screen; as the effluent overflows the tank and runs down over the full width of the screen, liquid passes through the screen openings, leaving solids behind on the screen. The solids

are washed downwards and drop onto a storage and draining pad. The lack of moving parts means maintenance and power requirements are low. However, regular washdown is necessary, and acid-washing to remove struvite may be needed to prevent blinding of the screen. A wash with disinfectant may be necessary if a biological film begins forming.

Inclined stationary screens are suited to a higher-solids-content effluent than sedimentation basins are suited to, but are limited to a TS content, in inflow, of <5% (Zhang and Westerman 1997). Separation efficiency (the capacity of the system to separate effluent into a 'solid' fraction and a liquid fraction) must be not be considered in isolation from the recovered solids content, as high separation efficiencies can be obtained with larger screen openings producing 'solids' with a dry matter content of <5%; that is, still liquid. Separation efficiencies of 20% to 30% TS and up to 10% N and 15% P are possible when solids with a dry matter content of 12% to 23% are produced.

Elevating stationary screens

Elevating stationary screens, or flighted conveyor screens, have a narrow inclined screen with its lower end in the effluent collection channel. A series of paddles move the effluent up the screen, allowing liquids to pass through the screen before discharging the remaining solids from the upper end onto a draining pad. Reported efficiencies are similar to that of inclined stationary screens. Although the elevating stationary screen overcomes the need for regular cleaning associated with the inclined stationary screen, it has a high maintenance requirement owing to its moving parts and to abrasion between paddles and screen.

Rotating screens

Rotating screens have a drum-type screen, the surface of which rotates past a fixed scraper to dislodge solids after the liquid drains through. Reported efficiencies are similar to that of inclined stationary screens.

Screw press separators

Screw press separators use a straight or tapered screw (auger) to compress solids within a perforated or slotted cylinder. Liquid is forced out through the screen openings by pump pressure and the rotating screw. Solids are pushed out the end of the barrel through an adjustable retainer.

Presses can operate at a higher TS content than can stationary inclined screens. Separation efficiency can be poor for dilute effluent but increases with solids concentration. Presses produce a drier solid than most mechanical devices—around 30% dry matter. Capital costs and power requirements are substantially higher than for stationary inclined screens.

Centrifuges

Limited results for centrifuges suggest good separation efficiency and dry matter content of recovered solids (>20% TS), but their use is limited by low throughput, high energy consumption and high capital expense.

Performance measures

The range in different types of solids separation systems available, not to mention suppliers, means that separation efficiency is a key criteria for comparing the performance offered by different types of separation systems. However, before doing so, it is important to note the following:

- Variability in particle size distribution (with manure type), TS content and flow rate make it difficult to draw anything other than general conclusions about the suitability of separator types for a particular farm. Results can vary greatly for any given device; take care in extrapolating results from published studies to individual farms.
- There is currently no standard for testing and reporting separator performance. Reported results must be scrutinised carefully to determine the method used to calculate efficiency, and whether the characteristics of the tested effluent are relevant. Published results come mainly from the USA, where rations are fed to housed or lot-fed cows and effluent may include readily separable organic bedding and waste feed.

Separation efficiency is the capacity of the system to separate effluent into a ‘solid’ fraction with high organic matter and nutrient concentrations and a liquid fraction with low concentrations. Solids separation systems will generally remove a good proportion of the total suspended solids (TSS) but only a small amount of total dissolved solids (TDS) with the separable suspended solids. As 62% to 83% of the TS is present as TSS (Loehr 1984), a TS separation efficiency of 60% is considered very good. Researchers have been attempting to increase the removal of solids by the use of chemical treatment to remove some of the TDS that comprises the remaining 15% to 40% of TS in raw effluent. Although separation efficiencies of 85% or more can be achieved by combining chemical treatment with gravity or mechanical approaches, the improved performance incurs a much higher cost (see chapter 5 ‘Odour emissions and control’).

Separation efficiency is usually reported using one of two measures: the reduction in concentration (approximate) or (dry) mass balance. The difference in the calculated efficiencies can be substantial and, if the method used is not specified, this omission can lead to costly misrepresentation of equipment suitability.

Reduction in concentration (approximate) method

This is a commonly used measure as it does not require the measurement of volumetric or wet-mass flow rates in influent, liquid or solids streams. Other parameters can be substituted for TS concentration (e.g. VS, TKN, TP) to calculate the respective separation efficiencies.

$$E_{approx} = \frac{C_{\inf luent} - C_{liquid}}{C_{\inf luent}} \qquad (1)$$

where E_{approx} = separation efficiency

$C_{influent}$ = concentration of TS (or VS, TKN, TP etc.) in influent

C_{liquid} = concentration in liquid or effluent fraction.

However, Equation 1 assumes an insignificant flow rate in the solids fraction and that the volumetric or wet-mass flow rate of the influent stream is equal to that of the liquid fraction. This is not necessarily correct and can lead to significant errors. Separation efficiency should therefore be calculated on a dry mass basis where possible.

Mass balance

An exact measure of separation efficiency can be calculated as:

$$E = \frac{(C_{\inf luent} \times Q_{\inf luent}) - (C_{liquid} \times Q_{liquid})}{C_{\inf luent} \times Q_{\inf luent}} \qquad (2)$$

By substituting $M = CQ$, where Q = wet mass or volumetric flow rate and M = dry mass:

$$E = \frac{M_{\inf luent} - M_{liquid}}{M_{\inf luent}} \tag{3}$$

or

$$E = \frac{M_{solids}}{M_{\inf luent}} \tag{4}$$

Note that if using the mass balance approach to analyse sediment basin performance, use Equations 2 or 3, not Equation 4. As the wet mass in the separated solids can be measured only upon clean-out, a proportion of the VS will decompose in the interim period, leading to a significant reduction in the mass of the solid fraction recovered.

Worked examples

Using a stationary inclined screen with 1.3-mm openings for flushed effluent with 0.7% solids, Wright (2005) recorded the results in Table 1.

Table 1. Example 1: Flushed effluent with 0.7% solids was passed through a stationary inclined screen with 1.3-mm openings.

	Influent	Liquid fraction	Solid fraction
Concentration (g TS L^{-1})	6.95	2.87	43.3
Flow rate ($L{\cdot}min^{-1}$)	11 130	10 000	1120
Dry mass ($kg{\cdot}min^{-1}$)	77.4	28.7	48.5

The calculated separation efficiency is good by either measure (E_{approx} = 59%, E_{mass} = 63%); however, it is important to recognise that the solids stream is still a liquid (4.3% TS, compared to raw manure at 9% to 13% TS) and remains difficult to handle. Although this particular separator and manure combination may be suitable for concentration before a second separation, it does not produce a material that can be handled as a solid.

Conversely, although a screw press separator with 1.3-mm screen openings gave a low separation efficiency (E_{approx} = 10%, E_{mass} = 13%), it produced material with 22% TS that can be handled as a semi-solid (Table 2) Burns and Moody (2003).

Table 2. Example 2: Scraped manure was diluted to 1% and put through a screw press separator.

	Influent	Liquid fraction	Solid fraction
Concentration (g TS L^{-1})	10	9	216
Flow rate ($kg{\cdot}min^{-1}$)	185	183.9	1.1
Dry mass ($kg{\cdot}min^{-1}$)	1.9	1.7	0.2

Performance reported in the literature

Appendix A presents a summary of performance results taken from a range of published research on the solid–liquid separation of dairy effluent. Table 3 summarises those results for general system planning. Note that significant departures from the values shown in Table 3 have been reported, so the data should be considered only as a starting point for system planning.

A number of other mechanical separators have been considered for solid–liquid separation of livestock manure; Ford and Flemming (2002) (pig, beef and dairy manure) and Watts et al. (2002a) (pig manure) are useful references for those.

2.1 Solid-liquid separation systems

Any farmer considering a mechanical separator should ask the prospective supplier or manufacturer for actual data from effluent with characteristics similar to their own. If data are not available, a trial run based on a large sample of the farmer's effluent should be performed before system design and installation. Compare the data provided against the performance reported in Appendix A.

Table 3. Suggested separation efficiencies for initial system planning.

Separator [1]	TS (%)	VS (%)	N (%)	P (%)	K (%)	Dry matter (%)
Trafficable solids trap	50	55	30	35	15	19
Stationary inclined screen	25	25	10	15	5	18
Screw press	20	20	5	5	0	30
Screw press (pre-concentrated to 10% TS)	60	65	25	25	10	30

[1] All effluents assumed to have typical TS concentration of <1% unless otherwise noted.

Note that, along with performance, the choice of separation system must also consider capital and ongoing operation and maintenance costs, reliability, skill required and expected service life. No dairy-specific data on these important issues were available, although some detailed information on pig manure exists (Watts *et al.* 2002b).

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2.2 Direct application systems

Effluent management systems that rely on the direct application of effluent to pasture are forced to operate even when conditions don't suit irrigation, such as during periods of rainfall or slow pasture growth. However, direct application may be necessary in situations where storage ponds introduce additional environmental risks.

Direct distribution versus storage

An early and sometimes contentious decision that must be made in designing an effluent management system is the choice between 'direct' application of effluent from the collection sump to the reuse area and storage in an effluent pond before periodic drawdown and distribution. The decision can be contentious because each approach has difficult-to-quantify environmental risks that depend on the specific characteristics of each site. Such characteristics and the limitations they pose are summarised in Table 1.

Table 1. Site characteristics and issues to be addressed.

Characteristic	Direct application of effluent	System including effluent storage
Soil type	Soils with high infiltration rates are needed to minimise runoff following any applications made during wet weather (or when soil moisture deficit is limited)	Site requires *in situ* material suitable for the construction of relatively impermeable clay liner (or a membrane liner must be installed)—see chapter 2.4 'Pond site investigation'
Vulnerable groundwater	Care must be taken to avoid over-application of nutrients resulting from the limited area of coverage and poor uniformity normally associated with such systems	As above
Topography and proximity to watercourses	Runoff resulting from application during periods of wet weather or limited soil moisture deficit must not enter a watercourse	Site may pose some restrictions on the location of a pond, e.g. distance to watercourse, above flood height, availability of site with suitable slope
High-rainfall region	As above	Calculated storage requirement may be impractical
Proximity to neighbours	Runoff resulting from application during periods of wet weather or limited soil moisture deficit must not leave the property boundary	Odours from a poorly sited pond might affect neighbours
Application system	Must be capable of very low application rates and depth to avoid runoff when soil moisture deficit is limited	The ability of the system to handle entrained solids is an issue for single pond systems; application rate is limited by soil infiltration rate
Enterprise viability		The cost of extraordinary measures such as membrane liners may not be justified for small or marginal operations

In general, storage ponds are recommended, unless site-specific conditions prevent their use, as they provide the opportunity to defer effluent application until conditions are suitable for irrigation, which has been shown to almost eliminate nutrient loss from land-applied effluent (Houlbrooke *et al.* 2004).

Regional soil and groundwater considerations

Some dairy regions have hydrogeological settings that are characterised by highly permeable subsoils and vulnerable groundwater resources with a high beneficial use value (for example, the limestone soils of south-eastern South Australia). Direct application systems may need to be retained as an option for small farms that cannot justify a long-term investment in pond liners if the risk of groundwater contamination from an ineffectively sealed pond constitutes a larger risk than applying effluent directly. Provided application rates are well managed, the direct application of effluent to the highly permeable soils represents a smaller nitrogen load than urine patches.

Management of daily application systems

Where a daily application system is currently being used without problems or can be justified as the best alternative for the farm, the following measures should also be adopted:

- Fit rainwater diversion devices and adopt effluent minimisation strategies where possible (see chapter 2.6 'Effluent storage requirement').
- Install an effective solids separation trap to reduce the nutrient loading on the reuse area, the likelihood of equipment blockages and excessive wear.
- Give the sump sufficient buffer storage (see Chapter 1.5 'Sump design') to avoid irrigation during rain. Even following a solids separation trap, the settlement of solids that will occur if storage exceeds 30 min means that agitation equipment will be needed.
- Select the reuse areas with the assistance of a tool such as the farm nutrient loss index (FNLI) (Melland *et al.* 2007). Locate them as far as possible from waters, avoiding steep terrain or topography that concentrates runoff.
- Select irrigation equipment with a low application rate, large irrigated area and high coefficient of uniformity.
- Develop and implement backup or contingency plans in the event of pump failure or equipment breakdown.

Risk of contaminant movement off-site

It is difficult to quantify the risk of nutrient loss in runoff and drainage resulting from direct application systems without detailed monitoring. Fyfe (2004) investigated the export of contaminants from a direct land application system that used a travelling irrigator in the Southern Highlands of NSW and concluded that the system did not provide adequate control of nutrients in wet weather even under recommended nutrient loading rates. Nutrients were exported from the reuse site (unfertilised, grazed) in greater quantities than from the control site (fertilised, grazed): 20.6 $mg{\cdot}L^{-1}$ TKN, 8.6 $mg{\cdot}L^{-1}$ NH_3-N, 7.8 $mg{\cdot}L^{-1}$ TP, 6.4 $mg{\cdot}L^{-1}$ dissolved reactive phosphorus (DRP) in runoff from the reuse site versus 3.4 $mg{\cdot}L^{-1}$ TKN, 0.8 $mg{\cdot}L^{-1}$ NH_3-N, 0.9 $mg{\cdot}L^{-1}$ TP, 0.7 $mg{\cdot}L^{-1}$ DRP from the untreated control. Variation in concentrations between sampling events suggested that the 'nutrient losses were not governed by soil interactions, but were a result of direct wash-off of waste'.

Houlbrooke et al. (2004) reported that nutrient losses from a single badly managed irrigation can be significant. Similarly, Misselbrook et al. (1995) found that 'losses of nutrients were higher following applications made with the soil at field capacity and rainfall soon after application'.

In general, contaminated runoff is more likely to move off-site on steeply sloping land, less permeable gently sloping land, or land that receives runoff from higher ground (Fyfe 2004). However, it is difficult to set criteria to determine whether a site is suitable

for direct application as, apart from topography, the likelihood of contaminated runoff loss is influenced by dynamic factors such as soil moisture deficit, soil permeability, amount and intensity of rainfall, and amount of effluent applied. As the time interval between the application of effluent and any subsequent rainfall is one of the most important factors governing the concentration of nutrients in runoff (Misselbrook *et al.* 1995), environmental authorities in all states recommend pond systems with sufficient capacity to store effluent during times of limited moisture deficit .

Management plans and monitoring

Farmers adopting direct application systems must therefore be prepared to provide sufficient information to regulatory authorities to justify their choice over a system with wet weather storage. Indeed, farmers in south-eastern SA are required to produce an Irrigation Management Plan (IMP) if effluent is applied during that part of the year when average rainfall exceeds average evaporation (Rural Solutions SA 2005). The intent of the IMP is to prevent nutrients, particularly nitrogen, from leaching past the root zone and entering groundwater, and to monitor for any impacts on ground or surface waters and soil. Although the broader adoption of such requirements is not currently being advocated, the process of thorough planning and follow-up monitoring is commendable.

Farm nutrient loss index

One tool that may offer some assistance in choosing between potential reuse areas is the farm nutrient loss index (FNLI). Although the FNLI is also relevant to selecting reuse areas for systems with storage ponds, the increased risk of runoff under direct application in wet weather or to soils with limited moisture deficit suggests that additional emphasis on site selection issues is warranted.

The FNLI allows a qualitative assessment of the risk of nutrient loss to the environment in all of the dairying regions in Australia (Melland *et al.* 2004). It determines the risk of N and P loss at the paddock scale and allows the user to evaluate the broad effect of different management practices on nutrient use efficiency while minimising environmental impacts. It does not estimate actual loss; rather, it assesses the risk of loss from a paddock or management unit in an average year.

The FNLI uses easily quantifiable inputs grouped into source factors and transport factors. The options for each factor are assigned a rating of 1, 2, 4 or 8, based on their potential to increase the risk of nutrient loss. As the relative importance of each factor varies between grazing regions, the ratings are weighted by a multiplier before being summed to determine overall risk. Risk ranking categories (low, medium, high or very high) are based on validation against field data. For example, Table 2 shows the options for the 'Effluent application and timing' factor. Melland *et al.* (2007) describes all of the other factors that are important when considering the risk of nutrient loss.

Table 2. Factor assessment criteria and ratings—'Effluent application and timing' (Melland et al. 2007)**.**

Rating	1	2	4	8
Effluent application and timing	Summer or autumn surface application or incorporation. Back-up recycle dam captures excess flood-irrigated effluent	Spring application when no heavy rain forecast for 7 days. 'Short watering' used to eliminate runoff from flood irrigation	Effluent applied when soils already waterlogged or heavy rain expected in <7 days	Effluent applied to land during winter, or no effluent storage system. Effluent drains directly off-farm

The FNLI will not provide a determination in the choice between a direct application system and one with a storage pond. However, where a farm has an existing direct

application system or intends to implement one, it will provide some assistance in determining on which paddocks effluent may be applied to minimise risks.

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2.3 Anaerobic, aerobic and facultative ponds

Ponds may be designed to reduce organic, nutrient and pathogen loadings in effluent, thus producing an effluent more suitable for reuse than raw effluent. Ponds do not provide a means for disposal of effluent as the pollution potential of the effluent leaving the pond is still too high for discharge to waters. More importantly, well managed ponds provide a means of storing effluent produced during periods when direct application may result in runoff.

When operating correctly, ponds can remove 95% of BOD and reduce the concentration of nutrients and pathogens in effluent. However, poorly designed and managed ponds can result in problems such as groundwater pollution, overtopping and spills, rapid sludge build-up, excessive crusting and unacceptable odour emissions.

Storage versus treatment

The terminology used to describe and differentiate between pond systems is sometimes misused. The most important function that a pond provides is containment; that is, providing sufficient storage to avoid having to distribute effluent during wet weather (see chapter 2.6 ‘Effluent storage requirement’). Indeed, if wet weather storage is the sole objective of the pond system, a single pond (in conjunction with a trafficable solids trap) is often sufficient for smaller farms and offers lower nutrient (particularly nitrogen) losses before reuse.

However, ponds are often adopted to improve effluent treatment where a farmer intends to:

- reduce odour during and after land application
- recycle effluent for flushing yards and lanes
- reduce the likelihood of blockages in conventional irrigation systems
- reduce nutrient and pathogen loads in effluent
- produce biogas.

Although single ponds can be designed to provide both treatment and storage, their efficacy is limited by effluent short-circuiting from inlet to outlet (see section ‘Inlet and outlet structures’ in this chapter). For the purposes of this document, treatment pond systems comprise two (or more) ponds in series: usually an anaerobic pond followed by a facultative pond that provides the storage capacity (note that generally only the final pond provides storage). Variations on this arrangement include three ponds (anaerobic pond followed by separate facultative and storage ponds) and dual anaerobic ponds (in parallel) to enable off-line desludging on a regular basis.

Settlement and biological treatment processes

With typically long detention times (i.e. weeks to months), settling is responsible for the removal of the majority of suspended solids and organic nutrients entering anaerobic ponds (Reed *et al.* 1995). Gravitational settling can account for removal rates of >50% for TS and VS, and >30% for N and P (see chapter 2.1 ‘Solid-liquid separation systems’).

Whether the organic matter is deposited in the settled solids or remains in suspension, it is decomposed by bacteria. Ponds contain extremely large numbers of bacteria, which use the effluent as an energy source for growth. The oxygen requirements of the bacteria and their relative numbers determine the classification of the pond as either anaerobic (absence of oxygen) or aerobic (measurable dissolved oxygen present). In practice, most ‘aerobic’ or storage ponds have anaerobic conditions at depth and may

be more appropriately termed ‘facultative’ ponds, containing a mix of anaerobic, aerobic and facultative bacteria, which can grow with or without oxygen. A comprehensive review of the biological communities found in animal effluent treatment ponds is provided by Hamilton *et al*. (2006).

Anaerobic ponds

Anaerobic bacteria occur in the intestinal tract of ruminants and do not need free oxygen to survive. Conditions in an anaerobic pond allow such bacteria to continue decomposing the remaining organic compounds in the manure (polysaccharides, proteins, fats), producing methane and carbon dioxide. Anaerobic bacteria are present throughout most of the water column, but activity is concentrated in the layer immediately above the sludge.

Anaerobic decomposition is a three-stage process—hydrolysis, fermentation (or acidogenesis) and methane formation (or methanogenesis)—with different groups of bacteria involved in each. In hydrolysis, solid material is broken down by enzymes into soluble molecules. During fermentation, the soluble molecules are degraded by acid-former bacteria into acetate, hydrogen and CO_2. Finally, two groups of methanogens produce methane from either acetate or hydrogen plus CO_2.

More detailed descriptions of the anaerobic process may be found in various texts, including Shilton (2005) and Metcalf & Eddy (2003), but the following points are important:

- The acid-formers produce volatile acids and other products which can cause objectionable odours if the methane-formers do not metabolise them.
- Anaerobic processes are sensitive to pH (methanogen activity is limited below 6.8) and to inhibitory substances such as ammonia, sulphide, copper, zinc and alkaline salts (see section ‘Pond management’ in this chapter for concentrations).
- The methane-formers have very slow growth rates, with a doubling time of days compared with hours for the acid formers. Large increases in the organic loading rate that exceed the capacity of the methane formers to complete the stabilisation of the fermentation products may cause incomplete anaerobic decomposition with increased odour emissions the likely result.

Recommended loading rates

Recommendations for anaerobic pond loading rates in Australian animal agriculture have traditionally been based on the Rational Design Standard proposed by Barth (1985). Following observations at four functional dairy lagoons in South Carolina, USA, Barth proposed a maximum volatile-solids loading rate ($VSLR_{max}$) of 0.17 kg VS m^{-3}·day^{-1} for dairy effluent (compared with 0.10 kg VS m^{-3}·day^{-1} for pigs and poultry). Subsequent US design standards removed the distinction between animal species and adopted a $VSLR_{max}$ of 0.085 VS m^{-3}·day^{-1} (ASAE 2004) to 0.10 kg VS m^{-3}·day^{-1} (USDA-NRCS 1996) for all uncovered anaerobic ponds.

The relationship between loading rate and odour emissions is an important design concern. Recent research suggests that anaerobic ponds that would be normally be considered undersized (having a high organic loading rate) can operate satisfactorily (Skerman 2007). Odour emission data from the pig industry suggest that higher pond loading rates may enable total odour emissions to be reduced at the planning stage via a decrease in the required pond volume and, consequently, surface area (see chapter 5 ‘Odour emissions and control’). Smaller ponds may also offer advantages in reducing construction and desludging costs (excavators and agitators have a limited reach). In addition, as covers will become more common for the control or capture of odours and greenhouse gases (GHGs) in the future, cover costs will also be minimised by the smaller surface area. Skerman (2007) identified that loading rates at least twice the

current $VSLR_{max}$ may minimise odour emissions (based on a 10-year desludging period), so there are significant potential advantages to be gained from optimising the current design criteria.

Research in progress for the pork industry (APL project no. 2108) is currently trying to establish design criteria for highly loaded ponds in an effort to minimise system costs and odours. Preliminary findings suggest that highly loaded ponds with VS loading rates of 5 to 10 times that suggested by Barth (1985) function effectively. Sludge management will be critical in such highly loaded systems.

The propensity of dairy effluent to form a crust is an important point of difference from piggery effluent, but research is yet to identify the factors involved in crust formation and the loading rate at which the crust becomes excessive and causes operational problems. Unfortunately, little data is available for designing ponds with a stable, but not excessively thick, crust. Misselbrook *et al*. (2005) suggest that ‘crust development occurs as a result of solids in suspension in the stored slurry being carried to the surface by bubbles of gas (carbon dioxide, methane) generated by microbial degradation of the organic matter. Evaporation at the surface will promote drying and binding of the particles at the slurry surface, forming a crust.’ Both the total solids (TS) concentration and the nature of the solids in the effluent appear to be important. Misselbrook *et al*. (2005) report that no crust formed on ‘slurries’ with a TS content of <1%. Environmental factors (temperature, wind speed, solar radiation, rainfall) that influence surface drying also appear to be important, as a ‘robust’ crust becomes evident only after at least 250 mm of evaporation occurs. As most effluents from Australian dairies have a TS < 1% and many form crusts, it would appear that the mechanisms of crust formation still require more local research.

Similarly, there has not been any attempt to identify a $VSLR_{max}$ suitable for the dairy industry under Australian conditions, where dilute effluents are the norm (TS < 1%). Extension guidelines for dairy effluent ponds have traditionally assumed that the formation of a crust is considered to be a sign of overloading. However, considering the potential for a crust to reduce odour and GHG emissions (see chapters 5 ‘Odour emissions and control’ and 8.2 ‘Greenhouse gas emissions’), that view may need to be revised. Unless the crust is causing blockages in transfer pipes, the benefits in leaving the crust intact are significant: a physical barrier to gas transfer, maintenance of anaerobic conditions, oxidation of odour and GHG emissions. Misselbrook *et al*. (2005) identified that crusts on slurry storage tanks reduce ammonia emissions by 50%.

Without additional research on dilute dairy effluent, it is premature to suggest any large increase in the recommended $VSLR_{max}$. US guidelines (USDA-NRCS 2003) retain a $VSLR_{max}$ of 0.17 kg VS $m^{-3}\cdot day^{-1}$ for anaerobic lagoons with impermeable covers (see chapter 8.1 ‘Production and beneficial use of methane’). Given the need to avoid problems caused by excessive crusting under a cover, a $VSLR_{max}$ of 0.17 VS $m^{-3}\cdot day^{-1}$ is appropriate for all ponds and should be used for the design of anaerobic ponds in the Australian dairy industry until more definitive data are available.

Regional adjustments to $VSLR_{max}$

As bacterial growth and the resulting rate of decomposition of organic matter slow with decreasing temperatures, $VSLR_{max}$ is usually adjusted for regions with different temperature profiles. A pond activity ratio (K) has traditionally been used to adjust $VSLR_{max}$ to the design VSLR for a particular site. Figure 1 shows lines of equal K (‘iso-K’) values across Australia (Kruger *et al.* 1995).

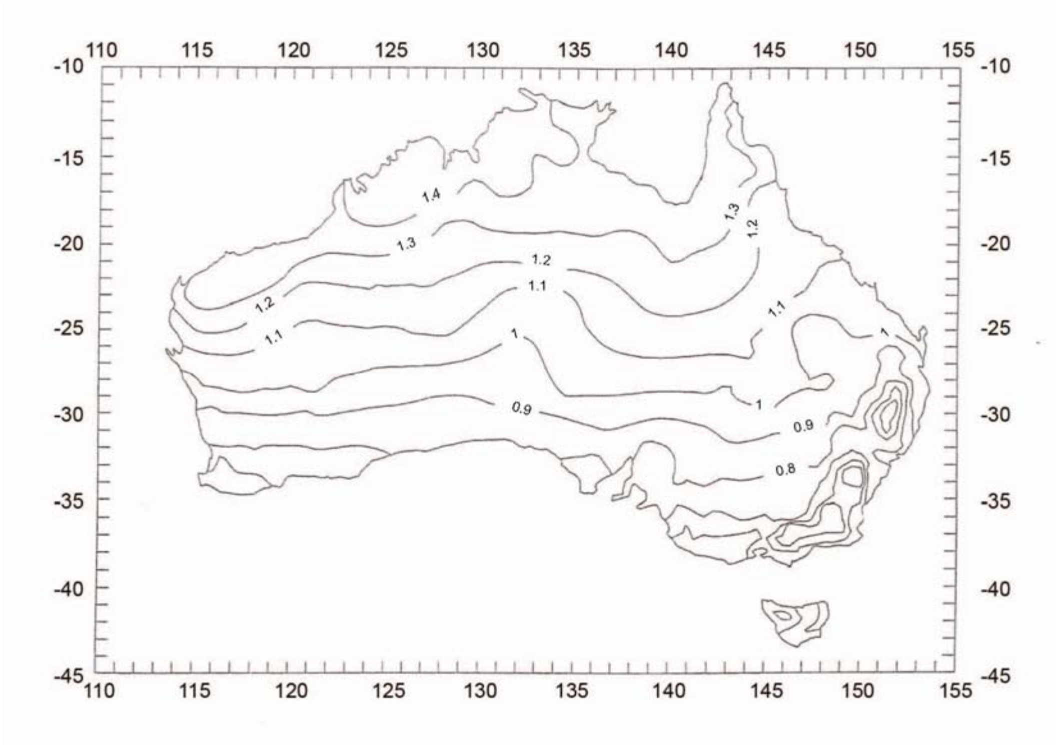

Figure 1. Map of iso-*K* lines in Australia used to adjust $VSLR_{max}$ (Kruger et al. 1995).

Recommended anaerobic pond design criteria

The design volume of the anaerobic pond ($V_{anaerobic}$) is the sum of the minimum active treatment volume (V_{active}) and the volume of sludge accumulation (V_{sludge}) expected over the selected desludging period:

$$V_{anaerobic} = V_{active} + V_{sludge} \tag{1}$$

Minimum treatment volume (V_{active})

The minimum treatment volume is based on the $VSLR_{max}$ recommended by (USDA-NRCS 2003) and the appropriate pond activity ratio, *K* (Figure 1):

$$V_{active} = \frac{TVS}{VSLR_{max} \times K} \tag{2}$$

where V_{active} = minimum active treatment volume (m^3)

TVS = total daily volatile solids load (kg VS day^{-1})

$VSLR_{max}$ = 0.17 kg VS $m^{-3}\cdot day^{-1}$.

Any future research attempting to identify $VSLR_{max}$ for dairy effluent should also investigate whether *K* remains relevant where odour is no longer a key design criterion, such as where emissions are limited by crusting resulting from higher loading rates.

Sludge allowance volume (V_{sludge})

Not all of the solids entering the pond are degradable: these non-degradable solids are referred to as fixed solids (see chapter 1.1 'Physical, biological and chemical components of effluent and manure'). In addition, some of the volatile solids degrade so slowly that they accumulate as sludge (~40% of VS added according to Chastain

(2006)). Although anaerobic decomposition continues in the ‘active’ sludge layer overlying the inert sludge, once the volume of inert sludge accumulates to the point where it reduces the minimum active treatment volume required for anaerobic digestion, the pond will not function satisfactorily, and desludging will be required, or increased odour emissions and solids carry-over will result.

ASAE Standards (ASAE 2004) recommend using a sludge accumulation rate of 0.00455 $m^3 \cdot kg^{-1}$ TS added for calculating the sludge allowance volume. This estimate is based on research by Barth and Kroes (1985) on three dairy lagoons in South Carolina, and that although it does not make an allowance for soil entering the pond from laneways via the cows’ feet, it appears to be conservative, overestimating the rate of sludge build-up according to Chastain (2006). Although there may be scope to reduce the sludge accumulation rate and the resulting pond size for many operations, overcorrection may result in unacceptably frequent desludging operations, so research data generated under Australian conditions should be obtained first.

The sludge allowance volume (V_{sludge}) is calculated as:

$$V_{sludge} = 0.00455 \times TS \times DP \times 365 \quad (3)$$

where V_{sludge} = sludge allowance volume (m^3)

TS = total solids loading ($kg \cdot day^{-1}$)

DP = desludging period (years).

Effluent management systems designed for freestall cow accommodation should include the additional contribution to sludge accumulation resulting from the portion of sand or organic bedding not removed by solids separation (see chapter 2.1 ‘Solid–liquid separation systems’).

Disposal of milk to an anaerobic pond

Disposal of waste milk to ponds is an appropriate strategy, though other options that offer some benefit (fed to calves, provided to pet food or stock feed manufacturers or pig farms) should be considered first. Alternatively, milk may be diluted with 6 to 7 parts of water for every part of milk (to achieve a 10 to 12 mm application with no more than 1500 kg BOD ha^{-1}) and applied directly to pasture.

Large slugs of organic loading may cause a temporary imbalance in pond function and result in increased odour emissions (see previous section ‘Anaerobic ponds’ in this chapter). For this reason, most guidelines suggest that no more than 2 days’ supply of milk can be added to well-functioning ponds without adverse effect, although this would be a rare occurrence. In the event of flooding preventing regular milk pick-up, however, pond disposal would be necessary.

Disposal of milk in the event of a bio-security scare must be discussed with regulatory authorities.

Size of anaerobic ponds

The anaerobic pond should be as deep as possible without reaching groundwater and have a minimum active depth (above the inert sludge layer) of 2 m remaining at the end of the design desludging period (Hamilton *et al.* 2006). Deep ponds offer a smaller surface area, resulting in lower oxygen transfer, less precipitation in wet climates, and less evaporation and salt build-up in dry climates; and a more stable temperature, improving the performance of the methanogens.

Some references suggest adopting a length-to-width ratio of not more than 4:1 (ASAE 2004) to allow more complete mixing, thus improving contact between the microbial population and the influent. It is possible that in extreme cases, a long, narrow pond may experience organic overloading at the inlet end (Shilton 2005). However, it may be

preferable to trade off a small reduction in anaerobic performance if desludging operations could be simplified by constructing ponds with widths that are manageable by agitators and excavators. As long-reach excavators have a maximum reach of around 18 m, adopting a maximum pond width of no more than 35 m would overcome some of the problems encountered when desludging is required. At this width, the corresponding maximum pond depths (at sites where there is no shallow groundwater) would be ~5.5 m with 3:1 batters and ~6.5 m with 2.5:1 batters; both are sufficient for the majority of anaerobic ponds being built.

Research by Shilton and Harrison (2003) suggests that a length-to-width ratio of 3:1 or more may improve treatment performance by reducing short-circuiting (see section 'Inlet and outlet structures' in this chapter).

For a detailed description of other design issues (freeboard, batters etc.), see chapter 2.5 'Pond design and construction'.

Aerobic ponds

In aerobic treatment ponds, aerobic microorganisms use dissolved oxygen to degrade the organic matter into carbon dioxide, water and cell biomass. Passive or naturally aerated ponds rely on oxygen produced by phytoplankton during photosynthesis and, to a lesser extent, diffusion of oxygen from the air into surface layers (Shilton 2005). Microorganism growth is rapid, and a large proportion of the organic matter is converted into cell biomass (which may also need to be treated and stabilised before the reuse of recovered sludge).

Naturally aerated facultative ponds are suited to relatively dilute effluents and should be used only after an anaerobic pond has provided substantial treatment. Although they could be used as a standalone option, the required surface area would be too large to be economical, and poor water quality would restrict light transmittance and algal photosynthesis.

See chapter 5 'Odour emissions and control' for details of mechanical aeration.

Light penetration and photosynthetic activity may extend down only 5 to 15 cm (the 'euphotic' depth) into typical dairy treatment ponds (Sukias *et al.* 2001). As algal growth is restricted in ponds where the mixing depth exceeds 5 times the euphotic depth, aerobic processes may be restricted below a depth of 75 cm. However, where the pond depth is <1 m, bottom-growing weeds may become established, decreasing capacity and, when decaying, adding biological load. The recommended depth for aerobic ponds is therefore a compromise between efficacy and practicality, and usually ranges from 1 to 1.5 m.

True aerobic ponds are rare in agricultural effluent treatment systems, as many so-called 'aerobic' ponds have anaerobic conditions below the top 20 cm (Sukias *et al.* 2001). Fortunately, aerobic ponds are not necessary, as reuse of agricultural effluent is the most suitable option, and facultative ponds offer a more practical option. Facultative ponds can maintain an aerobic surface layer for odour control and, being deeper than aerobic ponds, minimise the footprint required to provide sufficient storage capacity.

Facultative ponds

Facultative ponds are those in which a combination of anaerobic, aerobic and facultative (able to grow in either the presence or absence of oxygen) bacteria stabilise effluents.

Standard surface loading rates are based on biological oxygen demand (BOD) rather than VS; 'aerobic' loading rates of 20 to 50 kg BOD $ha^{-1}·day^{-1}$ (Wrigley 1994) have typically been used for design. In contrast, the New Zealand dairy industry uses a loading rate of 84 kg BOD $ha^{-1}·day^{-1}$ (Dairying and the Environment Committee 2006), even though average temperatures are lower than in most Australian dairy regions.

2.3 Anaerobic, aerobic and facultative ponds

Bacterial activity depends on temperature, so loading rates should be tailored to suit each climatic region. To this end, Reed *et al*. (1995) suggested ranges of 22 to 45 kg BOD $ha^{-1}\cdot day^{-1}$ (for sewage) where the winter air temperature is 0 to 15 °C, and 45 to 90 kg BOD $ha^{-1}\cdot day^{-1}$ where the winter air temperature is >15 °C. This approach results in loading rates of 30 to 50 kg BOD $ha^{-1}\cdot day^{-1}$ across Australian dairy regions, in agreement with the loading rates suggested by Wrigley (1994). Figure 2 shows mean temperatures for June to August across Australia based on data from 1950 to 2005.

Facultative ponds are typically designed with a depth of up to 2.5 m (Metcalf & Eddy Inc. 2003). However, effluent 'reservoirs' with depths exceeding 6 m are used to store sewage effluent in several countries, and a loading rate of 50 kg BOD $ha^{-1}\cdot day^{-1}$ is typically considered to be the maximum allowable loading if odour control is the goal (Shilton 2005). Therefore, facultative ponds may be deeper than 2.5 m to achieve the storage requirement if the loading rates of Reed *et al.* (1995) are adopted.

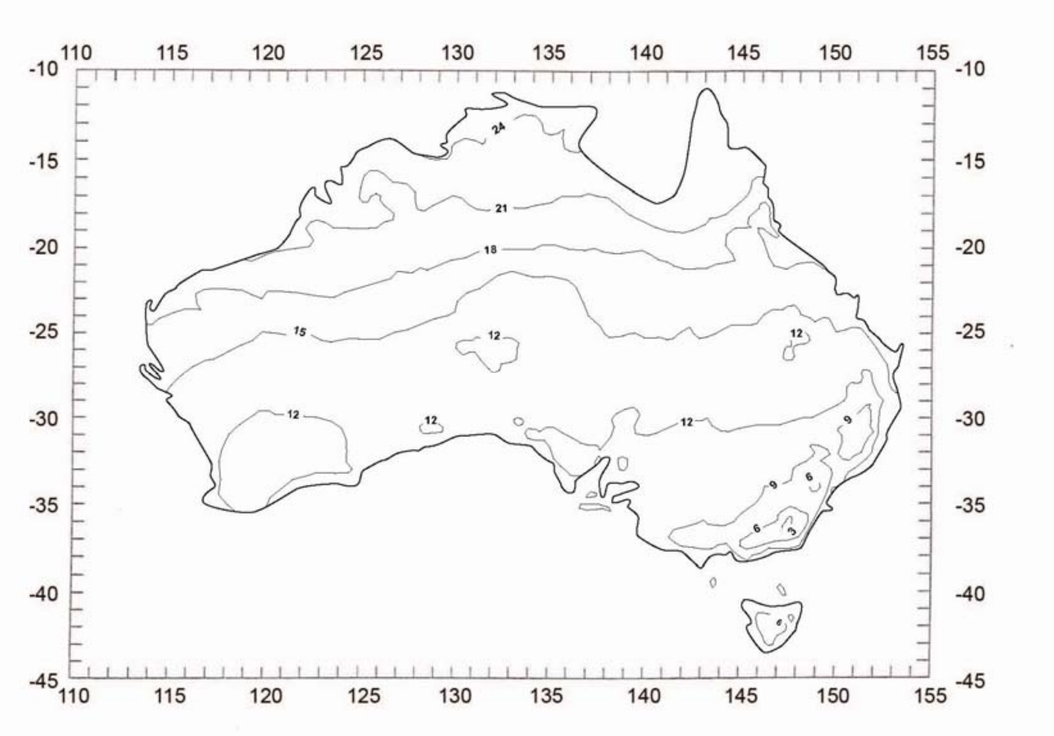

Figure 2. Mean winter temperatures in Australia (source—BOM).

Treatment pond performance

BOD

When sized appropriately, anaerobic ponds routinely remove 70% of BOD load (Metcalf & Eddy Inc. 2003). Removal efficiencies of 80% to 90% have been recorded in anaerobic lagoons designed to New Zealand dairy industry guidelines (Mason 1997).

Facultative ponds should remove 80% of incoming BOD; Mason (1997) confirms that that level of performance is possible. However, Sukias *et al.* (2001) found that facultative ponds in New Zealand typically removed only 40% to 50% of the BOD remaining in effluent after treatment in an anaerobic pond. When combined with the reductions achieved by the anaerobic pond, the pairing of an anaerobic pond and facultative storage pond removes around 90% to 95% of BOD.

VS

Chastain (2006) suggests that anaerobic dairy lagoons remove around 56% of VS load via settling. Pre-treatment by solid–liquid separation would remove some of the readily settleable solids before the effluent enters the anaerobic pond and therefore reduce the separation achieved in the pond.

Nutrients

DPI (2005) and the NSW Dairy Effluent Subcommittee (1999) both provide Table 1 as a guide to the fate of nutrients in pond systems, but neither provides a reference for the source of the data.

Table 1. Locations and losses of nutrient in ponds.

	N (%)	P (%)	K(%)
Effluent	30	40	90
Sludge	20	60	10
Loss by volatilisation	50	–	–

‘Typical’ characteristics of effluent

A table of ‘typical’ analyses of effluent from facultative storage ponds is provided for background information (Table 2). The inconsistent nature of dairy effluent means that standardisation using typical concentrations, as used for sewage effluent, is not prudent, and reuse systems for farms must be designed on a case-by-case basis. Although it may be useful to compare actual analyses with the tabulated data, the large standard deviations recorded mean that it would take an extremely large departure from the mean to suggest the result may be unusual.

Table 2. Mean effluent pond concentrations (standard deviation in parentheses).

Parameter	Units	Storage and single ponds Qld (SE) [a] n = 18	Storage ponds Vic (Gippsland) n = 79	Single ponds Vic (Gippsland) n = 12	Single ponds Vic (northern) n = 20
TKN	$mg \cdot L^{-1}$	167 (148)			
Total N	$mg \cdot L^{-1}$	167 (148)	286 (268)	429 (267)	311 (209)
Total P	$mg \cdot L^{-1}$	36 (22)	107 (206)	113 (63)	86 (70)
K	$mg \cdot L^{-1}$	274 (299)	474 (447)	479 (184)	361 (256)
Total S	$mg \cdot L^{-1}$		58 (119)	112 (101)	
EC	$\mu S \cdot cm^{-1}$	3904 (2111)			3216 (2132)
pH	–	7.9 (0.6)			7.3 (0.5)
Cl^-	$mg \cdot L^{-1}$	234 (207)			
Ca^{2+}	$mg \cdot L^{-1}$	98 (51)			
Mg^{2+}	$mg \cdot L^{-1}$	103 (66)			
Na^+	$mg \cdot L^{-1}$	225 (168)			
SAR	–	3.7 (1.9)			

a: (Skerman *et al.* 2006).

Pond management

Salts and inhibition

Salinity levels in ponds will gradually increase over time as evaporation removes some of the water, thus concentrating the remaining salts. High concentrations of alkaline salts (Na^+, K^+, Ca^{2+}, Mg^{2+}) can inhibit bacterial activity and, eventually, become toxic, causing failure of the treatment system. Table 3 indicates the likely concentrations at which bacterial activity may be stimulated or inhibited. Concentrations listed as

moderately inhibitory may cause temporary upsets if introduced suddenly, but with acclimatisation by the bacteria, may not significantly retard the treatment process. Be aware that toxicity due to a specific cation may be reduced by the presence of one or more additional cations, and that specialist advice may be necessary to interpret pond chemistry.

Table 3. Stimulatory and inhibitory concentrations of inorganic compounds (McCarty 1964)**.**

Substance	Stimulatory ($mg \cdot L^{-1}$)	Moderately inhibitory ($mg \cdot L^{-1}$)	Strongly inhibitory ($mg \cdot L^{-1}$)
Na^{+}	100–200	3500–5500	8000
K^{+}	200–400	2500–4500	12000
Ca^{2+}	100–200	2500–4500	8000
Mg^{2+}	75–150	1000–1500	3000
NH_4^{+}		1500–3000	3000
Cu^{2+}			0.5 (soluble), 50–70 (total)
Zn^{2+}			1.0 (soluble)

USDA-NRCS (1996) suggests that if the total salt concentration is in the range of 2500 to 5000 $mg \cdot L^{-1}$ (3.9 to 7.8 $dS \cdot m^{-1}$), the pond should be diluted (with stormwater) following drawdown. Safley *et al*. (nd.) agree with the upper end of that range, suggesting that at an electrical conductivity (EC) above 8 $dS \cdot m^{-1}$, the pond requires dilution to avoid bacterial inhibition. As Waters (1999) observed EC levels averaging 4.7 $dS \cdot m^{-1}$ (2.8–7.8 $dS \cdot m^{-1}$) in Victoria, this parameter must be monitored and managed by the farmer.

Inlet and outlet structures

Effluent should be transferred from the anaerobic pond to the facultative storage via a baffled pipe designed to exclude solids carryover and blockage by any crust that may be present. A 150-mm uPVC pipe fitted with T-junction and cut-off collars is required (Wrigley 1994). An extension on the T-junction should draw water from a depth of at least 300 mm below the surface (Shilton 2005).

Inlets and outlets must be located to avoid short-circuiting and maximise hydraulic retention time (HRT). In practice, this is difficult without tracer studies; a comprehensive review of the issue is provided by Shilton and Harrison (2003). In general, for roughly square ponds, horizontal inlets can ‘drive’ the pond contents to circulate at much higher velocities than if the flow moved simply from inlet to outlet, and options such as vertical inlets, manifolds and baffles may be necessary to prevent significant short-circuiting. Outlet location is also important; ‘sheltered’ positions out of any circulatory current are preferable. Sheltered positions are usually found in the corners of square ponds, or if the pond is irregularly shaped, in the smaller part of the pond.

In ponds with a length-to-width ratio of 3 or more, the best option is to direct a horizontal inlet to discharge across the shortest dimension to create a series of counter-circulating currents that die out as momentum decreases with distance from the inlet (Figure 3).

2.3 Anaerobic, aerobic and facultative ponds

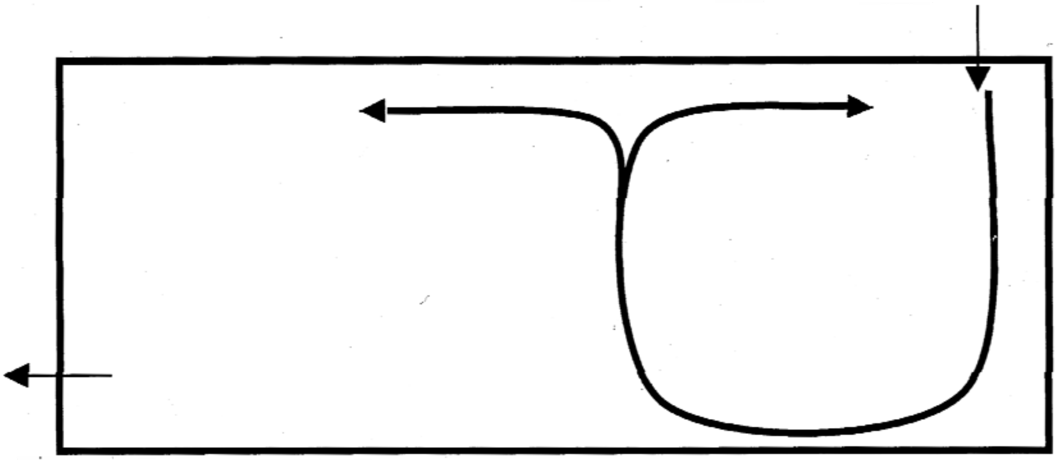

Figure 3. Circulatory currents in ponds with a length-to-width ratio of 3 or more (Shilton and Harrison 2003).

Maintenance requirements

As effluent ponds increase in size to accommodate increasing herd numbers and more intensive feeding strategies, so does the potential impact on the environment if those ponds are not maintained correctly and a spill or catastrophic failure occurs. Recommended maintenance practices include:

- maintaining well-grassed outside batters to avoid rilling erosion
- avoiding inside batter erosion where effluent enters ponds by extending inlet piping over or into water at least 1 m deep, or by stabilising the batter using rip-rap (a rock facing layer) or a lined entry 'gutter'
- checking for wave-action erosion of the inside batter (riprap may be necessary)
- preventing the establishment of (or removal of existing) shrubs and trees where their root systems may encroach upon embankments and create the potential for leakage through old root tracks or cracks
- feral animal control to eliminate any rabbit or fox burrows on or around the embankments.

Unfortunately, even good managers can fail to spot a slowly developing problem. Therefore, a regular inspection focused solely on maintenance is necessary. A signed and dated inspection record is also a useful tool for demonstrating good stewardship during any audit by regulatory authorities. An example of an inspection checklist can be found at http://www.lpes.org/Lessons/Lesson24/24_10_Lagoon_Checklist.pdf.

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2.4 Pond site investigation

Site investigations are critical for assessing the suitability of sites and soils for dairy effluent pond construction and ongoing maintenance, and the information garnered should be utilised in pond design. The primary objective is to provide secure storage that minimises seepage losses and facilitates recycling of treated wastewater. Secondary objectives include locating ponds to improve farm management and gaining earth for other farm infrastructure such as silage pits, feedpads and laneways.

Initial siting

Careful consideration is required in the siting of a dairy effluent pond, and a range of sites may need to be evaluated before the preferred location is chosen. It is unwise to site ponds solely on the basis of regulatory requirements or where historically effluent has flowed. As earthworks are typically required to achieve the effective collection of effluent and contaminated runoff, planning for a dairy effluent pond should be undertaken in conjunction with overall farm planning. The following points should be considered in initial pond site selection.

Farm integration

- Provide for effective and efficient collection of as much effluent and contaminated runoff from the dairy and other stock management areas as possible.
- Aim to achieve sufficient distribution of nutrients over the farm; this is important where gravity irrigation systems are used (McDonald 2006).
- Where required, allow for the recycling of treated wastewater for flood washdown.
- Integrate ponds with other farm features, such as using pond embankments as causeways, allowing for laneways, facilitating effective cow flow, improving all-weather access and making effective use of land while integrating the pond within the existing landscape.
- Provide for efficient pond operation, maintenance and monitoring; this is often achieved by ensuring that ponds are visible from the dairy.
- Prefer a site with all-weather access for maintenance of pumps and ponds.
- To limit the propensity for seepage, do not locate ponds adjacent to drains or incised water courses.

Environmental considerations

- Individual states have specific policies on the siting of effluent ponds in relation to flood waters and flood return periods; refer to these (Environment Protection Authority 2003). However, it is not recommended that these be definitively applied, as each site requires assessment on its own merits, and more sensitive areas may require stricter application of these policies than less sensitive areas. Ideally, avoid areas subject to 1-in-100-year floods. However, this is often not possible, so install adequate safeguards to protect the environment (see the next point).
- The NSW Dairy Effluent Subcommittee (1999) recommends protection of ponds against overtopping from 1-in-5-year floods in existing dairies, and from 1-in-25-year floods in new dairies.

2.4 Pond site investigation

- The soils at any proposed site need to undergo adequate soil geotechnical analysis (see 'Soli assessment') to determine suitability for pond construction (Standards Australia 1993).
- The potential for and the implications of both groundwater contamination from seepage and groundwater influx into the ponds should be assessed for any intended pond site as detailed below (Hopkins and Waters 1999).
- Acid sulphate soils must not be disturbed unless a Statement of Environmental Effect (ESS) or an Environmental Impact Statement (EIS) is produced (NSW Dairy Effluent Subcommittee 1999).
- Avoid sites which rely on the diversion of general catchment runoff, such as depressions and gullies, and do not divert flood flows elsewhere.

Regulatory requirements

Consultation with regulatory bodies is recommended to determine what is required before pond construction (e.g. a planning permit). Works need to adhere to any relevant State Planning Policy frameworks and local council permit requirements. Earthworks particularly attract regulation, given the potential danger of collapse or diversion of water. In Queensland, some water resource plans prohibit the taking of overland flow without an extraction licence. Farm dam legislation in Victoria is similar in intent.

Typically it is up to the applicant to demonstrate how their proposal fits with the relevant state government and local council policies. Land owners are expected to undertake sufficient planning to ensure that the system adopted is the best solution for both the operation and the surrounding environment.

Public amenity and food safety

The location of a dairy effluent pond should take into account the location of existing housing, other sensitive uses and land zoned for residential or urban purposes. In addition, effluent storage is not permitted within 45 m of a milk room in Victoria. Conventional practice favours the adoption of buffer distances based on the nature of a receptor and the organic loading generated by an emitter. However, in the case of dairy effluent ponds, buffer distances are often specified. Alternatively, proponents may wish to undertake odour dispersion modelling to demonstrate satisfactory performance for a proposed effluent pond (see chapter 5 'Odour emissions and control'). The general principle that should be adhered to is that no discharge should give rise to material detriment to any person (i.e. interfere with the normal use and enjoyment of life and property to more than a trivial or minor extent). It is critical to avoid alienation of neighbours. Whenever possible, they should be consulted as early as possible in the planning process. Ideally, existing buildings, topography and vegetation should be used to screen ponds from major roads and nearby residences.

Soil assessment

Soil investigations

It is imperative to investigate subsurface features of an intended site by test boring or excavation trenches and soil geotechnical analysis to determine the soil's suitability for pond construction. The results of this analysis combined with the specific intended use of the material (e.g. embankment core, outer embankment batter, embankment liner) should determine final suitability.

Although there is no standard for the number of investigation sites required, the number should depend on the footprint dimensions and on site soil and surface feature variability. It is critical that the subsurface investigations adequately represent the soils

across the proposed site. These investigations can be assisted by aerial photographs, farm plans containing topographical details or geophysical surveys (e.g. electromagnetic survey). Pertinent aspects to be assessed are detailed in Standards Australia (1993).

Soil physical assessment

Assess and document the exposed soil profile at each inspection site in detail and note features such as:

- depth of the each soil horizon (layer)
- texture
- colour and mottling
- structure
- mechanical resistance (hardness)
- friability
- porosity
- drainage status
- presence of natural lime ($CaCO_3$) or gypsum ($CaSO_4$)
- proportion and type of rock
- water table presence
- groundwater quality (EC)
- presence of sand or layers conductive to preferential water flow
- other distinguishing soil features; e.g. manganese layers, clay skins or structural peds.

For a full appreciation of the information that can be collected and definitions of soil-related terms, see McDonald *et al.* (1990).

Soil geotechnical analysis

Soil geotechnical laboratory testing is recommended for all pond sites and is required to determine permeability and structural stability (NSW Dairy Effluent Subcommittee 1999, Standards Australia 1993). The sampling and geotechnical analysis process should include the following considerations:

- Analysis of soil samples representative of material intended to be used in the pond floor, embankments and other pond structures.
- Analysis for dispersion, shrink–swell capacity, Atterberg limits and USCS engineering classification.
- Analysis for saturated hydraulic conductivity on remoulded samples to demonstrate that the material can achieve a permeability of $<1 \times 10^{-9}$ $m{\cdot}s^{-1}$ at the specified target density (Environment Protection Authority 2003, NSW Dairy Effluent Subcommittee 1999). However, as significant site variability is possible, more than one test may be required; the number should be based on recommendations from a qualified practitioner.
- AS 1289 (Standards Australia 2000) requires that the Electrical Conductivity (EC) of the test solution used to test saturated hydraulic conductivity be representative of the material. Although the NSW Dairy Effluent Subcommittee (1999) recommends using 0.01 M KCl as the test solution, either a sample of effluent or water with an EC adjusted to within the typical range (2.8–7.8 $dS{\cdot}m^{-1}$ according to Waters (1999) is suitable.

Soil suitability

The soil suitability information detailed in this section (from Skerman *et al.* (2004)) provides a useful indication of the suitability of site material for pond construction. See also AS 1726 (Standards Australia 1993).

The clay used to line ponds should be well-graded impervious material, classified as either CL, CI, CH, SC or GC (see Appendix A, Table A1, in AS 1726; (Standards Australia 1993). The lining material must conform with the particle size distribution and plasticity limits shown in Tables 1 and 2.

Table 1. Particle size distribution.

AS metric sieve size (mm)	Percentage passing (by dry weight)
75.0	100
19.0	70–100
2.36	40–100
0.075	25–90

Table 2. Plasticity limits on fines fraction, passing 0.425-mm sieve.

Liquid limit	30–60%
Plasticity index	>10%

If materials complying with these plasticity limits are not readily available, clays having liquid limits between 60% and 80% may be used as lining material, provided that the clay lining layer is covered with a layer of compacted gravel (or other approved material). The compacted gravel layer should have a minimum thickness of 100 mm to prevent the clay lining from drying out and cracking. To determine compliance with the above requirements, materials must be tested in accordance with AS 1289 (Standards Australia 2000). Any material which does not compact properly (e.g. topsoil, tree roots, organic matter) must not be used in clay lining or be placed in areas to be lined. Wherever non-dispersive materials are available, use them in preference to materials shown to be dispersive by the Emerson test, as described in Method 3.8.1 of AS 1289.

Groundwater

In addition to soil investigations, hydrogeological investigations are also important to determine the depth to groundwater and the location of geological formations that favour groundwater flow. Examine existing groundwater data to determine possible hydrogeological conditions and district beneficial groundwater use. These data can sometimes take the form of a risk assessment. The following important groundwater factors need to be considered:

- The potential for and the implications of both groundwater contamination from seepage and groundwater influx into ponds should be assessed.
- Individual states have specific policies on the siting of ponds in relation to the depth of groundwater below the pond base or below the natural surface. Although these policies need to be followed, each site requires assessment on its own merits, and final suitability will be dictated by pond depth, soil permeability and beneficial use of groundwater.
- Ponds should not be situated above groundwater resources that are deemed to be vulnerable to contamination unless those resources can be shown to be protected (NSW Dairy Effluent Subcommittee 1999). Karst limestone systems are at particular risk; these are omnipresent in south-eastern SA.
- Extra justification or safeguards are required where shallow or saline groundwaters occur (NSW Dairy Effluent Subcommittee 1999).

Final pond site suitability assessment

All relevant information should be assessed by a suitably qualified person to evaluate the appropriateness of a site for construction of a dairy effluent pond. Appropriate representative soil samples need to meet the suitability criteria of Skerman *et al.* (2004) as detailed above. In addition, information on flooding potential and groundwater also needs to be considered, along with public amenity and farm planning considerations, to determine site suitability.

The site assessment should result in recommendations on appropriate construction techniques and on monitoring and management. Further information is provided in chapters 2.5 ‘Pond design and construction’ and 7 ‘Monitoring and sampling’. It is advisable to have a written contract prepared before construction to assist the earthmoving contractor and to ensure that all parties agree on the nature of the works required and the costs likely to be incurred. Avoid verbal agreements.

Professional supervision may be necessary on larger, more sensitive sites or where materials are highly variable. It is advisable to determine whether any cultural and heritage overlays apply to the site (check with the local council) during the investigation and avoid any sensitive sites. In particular areas it may be necessary to undertake a cultural and heritage survey.

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2.5 Pond design and construction

Earthen pond design and construction must be based on the results of thorough soil and site assessments; see chapter 2.4 'Pond site investigation' for details. The design and construction of dairy effluent ponds should minimise the likelihood of seepage and pollution of groundwater or surface waters. Well defined standards of soil assessment and pond construction are necessary to ensure that the structural integrity of embankments, clay liners and associated pond features provide the level of security necessary.

Pond design and specifications

Pond sizing, shape and depth

For pond shape and depth, see chapter 2.3 'Anaerobic, aerobic and facultative ponds'. For storage requirements, see chapter 2.6 'Effluent storage requirement'.

Freeboard

Freeboard is the elevation difference between the full pond and the crest of the bank. Freeboard protects the bank from wave action, rilling, bywash flows and overtopping under high-intensity rainfall and fast filling. The freeboard of a pond needs to be specified for each situation to minimise the risk of overtopping. A minimum freeboard depth of 600 mm is mandated in SA (Environment Protection Authority 2003). Allow for 10% consolidation and compaction under stock or vehicle traffic.

Soil parameters

The propensity for seepage losses from earthen ponds is dictated by soil physical and chemical characteristics and the resulting hydraulic properties of the material forming the embankment, walls and floor of the pond. The soil characteristics of a site govern pond design and dictate the need for soil ameliorants and synthetic liners. Cohesionless soils provide poor embankment strength and cannot impound effluent. Heavy clays make suitable embankments and effluent storages but can crack when drying and are difficult to work when either wet or dry. Details of soil analyses and parameter thresholds are provided in chapter 2.4 'Pond site investigation'. Although not definitive, an indication of soil suitability is provided in Table 1. This is based on US Bureau of Reclamation (1977) and the Unified Soil Classification System (USCS) as per Standards Australia (1993).

Selective material placement

The various components of a pond and embankment should be zoned to allow appropriate materials to be used to the best advantage, providing for placement of specific soil layers in various zones of the pond. For example, dispersive soil layers could be confined to the floor, and layers consisting of natural lime or gypsum, which can enhance soil stability, could be used on the inner embankment.

Table 1. Rating of materials for embankments and pond floors.

Rating	Description	Percentage clay and silt and USCS rating	Linear shrinkage (%)	Emerson class No.
1 (Very good)	Very well graded coarse mixtures of sand, gravel and fines, D85 coarser than 50 mm, D50 coarser than 6 mm. If fines are cohesionless, not more than 20% finer than the No. 200 sieve.	Clay 10%–25%, silt + clay 20%–40%. These coarse-grained soils are generally suitable for use in earthworks. Such soils with a classification of GM of SM are not suitable for homogeneous or impervious zones of water storages. Soils are susceptible to tunnelling.	0–12	3, 7, 8
2 (Good)	(i) Well graded mixture of sand, gravel and clayey fines. D85 coarser than 25 mm. Fines consisting of inorganic clay (CL) with PI > 12. (ii) Highly plastic tough clay (CH) with PI > 20.	Clay 25%–40%. These generally fine-grained soils are suitable for most soil conservation earthworks. Pay attention to susceptibility to tunnelling and cracking.	12–15	2
3 (Fair)	Fairly well graded, gravelly, medium-to-coarse sand with cohesionless fines. D85 coarser than 6 mm; 0.5 mm < D50 < 3.0 mm. Not more than 25% finer than the No. 200 sieve.	Clay 10%–25%, silt + clay > 45%. These fine-grained silty soils have variable permeability and are likely to be erodible. Generally suitable only for upstream and downstream zones of a zoned embankment or in a modified homogeneous embankment with a filter zone.	17–22	4, 5
4 (Poor)	(i) Clay of low plasticity (CE and CL-ML) with little coarse fraction. PI 5–8. Liquid limit >25. (ii) Silts of medium to high plasticity (ML or MH) with little coarse fraction. PI > 10. (iii) Medium sand with cohesionless lines.	Clay > 40%. These fine-grained soils may leak if well aggregated, or may crack on drying if they have high-volume expansion or linear shrinkage values.	>22	6
5 (Very poor)	(i) Fine, uniform, cohesionless silty sand. C55 < 0.3 mm. (ii) Silt from medium plasticity to cohesionless (ML). PI < 10.	Clay < 10%, silt + clay < 20%. These coarse-grained soils are pervious and are not recommended for general use in homogeneous or impervious zones of water storages. Soils with a USCS rating of GC or SC may be suitable for water storage when well compacted at the optimum moisture content. A filter zone would be required.	None	Only clay soils can be dis-persive. Class 1 (Clay)

Source: US Bureau of Reclamation (1977).
PI: plasticity index.

Embankments

Pond embankments need to be constructed from appropriate material (see previous section) and appropriately compacted (see ‘Compaction’ below). Subsurface soil and geological conditions, identified in the site investigation, will dictate cut-off trench depth under an embankment. Although a minimum of 300 mm is recommended (NSW Dairy Effluent Subcommittee 1999), depths could range up to 2000 mm or, in special cases,

more. The depth is dictated by the depth and thickness of pervious strata underlying the proposed site.

Batter slope

The maximum batter slope should be 2.5:1 for internal and external walls and embankments where a bulldozer is used; 3:1 where a compacting roller is used (NSW Dairy Effluent Subcommittee 1999); and not steeper than 2:1 (DPI 2004). Flatter batters better maintain embankment stability, facilitate compaction, protect against wave run-up and improve safety in pond construction and maintenance.

Crest width

The embankment crest width should be a minimum of 3.0 m (DPI 2004); 3.0 to 4.0 m is preferable to allow vehicle access for construction and maintenance (Bradshaw 2002a, Bradshaw 2002b, DairyCatch 2006, NSW Dairy Effluent Subcommittee 1999). Crest width must allow for desludging activities, which generally require heavy machinery (EPA 2004). Some contractors prefer at least one of the long sides to be 6 m wide and to have approach and departure ramps with a slope of 1:10 to provide access for machinery during desludging.

Pond construction

In preparing a site for pond construction, before any land disturbance, put in place erosion and sedimentation controls to limit any off-site impacts. Strip and stockpile topsoil only immediately before construction. Consider and minimise any potential adverse impacts on adjoining sites from noise or dust.

During excavation, be alert for any material substantially different from that revealed in the pond site investigation and soil geotechnical analysis. If encountered, the differing material will need to be assessed, and pond design or construction practices must be adjusted. For example, the presence of excessively silty or sandy soil layers may require the construction of a compacted clay liner.

Although not mandatory and dependent on project scale, supervision of construction by a geotechnical engineer is recommended for quality control.

The objective of construction is not just to provide a hole. If a ramp or shallow batter can be installed as well, it will assist desludging and pumping out and improve escape prospects for stock and humans, reducing OH&S risks.

Floor and lining

The floor of a pond needs to be constructed from appropriate material and to be appropriately compacted (see next section). The soil geotechnical analysis will identify soils that crack excessively or contain <20% clay, and areas of exposed rock or sand, all of which will require lining. Where it cannot be demonstrated that, with conventional compaction, the *in situ* soil material can achieve a permeability of $<1.0 \times 10^{-9}\ \mathrm{m \cdot s^{-1}}$, a clay or synthetic liner must be installed on the floor and internal embankments. Documentation of permeability may be required (Environment Protection Authority 2003). A NATA-accredited laboratory should conduct these tests.

Pond lining—clay liners

Although it is impossible to achieve zero leaching through a compacted soil layer, leaching should be minimised to achieve a permeability of $<1.0 \times 10^{-9}\ \mathrm{m \cdot s^{-1}}$. Within effluent ponds, clay lining is typically required to achieve this. George *et al.* (1999) state that, in general, most studies show that the initial leakage rate of a storage pond is

high, but over time it is reduced as a result of sealing due to physical, chemical and biological processes within the pond. The flux of leachate through a pond liner is typically calculated by applying Darcy's Law under the assumption of saturated conditions (Tyner and Lee 2004). Ham (2002) found in a study of 20 animal effluent ponds that the hydraulic conductivity of compacted soil liners averaged 1.8×10^{-9} $m \cdot s^{-1}$, and found evidence that organic sludge moderated seepage rates. Cihan *et al.* (2006) developed a model to predict the formation of a seal by animal effluents through compacted soil liners, and found that over time, as the seal forms, infiltration rates are further reduced.

Where soil geotechnical results show that a clay liner needs to be laid on the pond floor and internal embankments, the following criteria apply:

- Permeability must be $\leq 1.0 \times 10^{-9}$ $m \cdot s^{-1}$.
- The material used should be classified as CL, CI, CH, SC or GC under the USCS (Skerman *et al.* 2004).
- Clay-dominant material should have a Liquid Limit between 30% and 60% and a Plasticity Index of >10% (Skerman *et al.* 2004).
- Clays with a liquid limit between 60% and 80% may be used as lining material provided the liner is protected from drying out by a minimum thickness of 100 mm of compacted gravel (Skerman *et al.* 2004).
- The liner must be 300 mm thick for a 2-m liquid depth and 450 mm for >2 m (Skerman *et al.* 2004).
- The pond should be filled with at least 500 mm of water upon completion to prevent the liner from drying out (Bradshaw 2002a).

Pond lining—synthetic liners

Where a synthetic liner is required, the following criteria apply:

- The liner must have permeability of ≤ 0.1 $mm \cdot day^{-1}$ and be installed to the manufacturer's specifications (Environment Protection Authority 2003).
- Seepage losses are usually associated with the joining of membranes, so take care with overlapping and welding of the liners.
- Protect the liner from desludging operations; for example, with a layer of used tyres (Bradshaw 2002a). Alternatively, use an agitator and pump for desludging ponds.
- Do not use synthetic liner where the water table is high: High water tables can place upward pressure on linings and cause damage.

Compaction

To attain sufficient compaction, specific geotechnical recommendations, based on soil geotechnical results, may be provided. If not, the following criteria will help to attain sufficient compaction:

- Where a bulldozer or excavator is used for construction, the floors and embankments must be appropriately compacted with a roller (NSW Dairy Effluent Subcommittee 1999).
- Scrapers or sheep's foot rollers give greater compaction rates.
- Compact soil with a moisture content within ±2% of optimum (Skerman *et al.* 2004).
- Where material is too dry to achieve satisfactory compaction, soil will have to be wetted.

- Each lift (a layer of fill generally not exceeding 200 mm) must be compacted to at least 95% of maximum dry density. This is typically achieved through eight passes of a sheep's foot roller (Skerman *et al.* 2004). Tamping foot and vibrating rollers can also be used.
- The lift thickness must not exceed 150 mm after compaction (Skerman *et al.* 2004).
- Construct liners with a minimum of two compacted lifts for optimum performance (Reinsch 2001).

Sealing

The sealing of a pond floor is aided by clogging by microbes and particulate matter (Magesan *et al.* 1999) and the formation of a sludge blanket (Silver *et al.* 2000). Pond design should be conducive to these processes.

Spillways, pipes and cut-off collars

In addition to stormwater diversion, ponds may require an emergency spillway for protection from severe storms; general dam design procedures may be used to determine the necessary dimensions.

Typically the weakest point in a pond is where pipes perforate embankments, so pipes through embankments should be avoided. However, a piped outlet to facilitate reuse is typically required; this should have seepage cut-off collars of 1200 mm × 1200 mm × 150 mm (Bradshaw 2002a, NSW Dairy Effluent Subcommittee 1999). Where concrete seepage cut-off collars are used, they should be reinforced with F81 mesh (NSW Dairy Effluent Subcommittee 1999). Where a suction pipe is used, this should be placed through the embankment berm at a height to assist pump priming (DairyCatch 2006). HDPE pipes should be used and be fitted with HDPE collars. Bentonite wafer collars can also be used.

Topsoiling

Topsoil should be stripped and stockpiled before construction, placed on external embankments after construction to a depth of between 200 and 300 mm (NSW Dairy Effluent Subcommittee 1999), and then grassed (Lower Murray Irrigation Action Group 1994) to stabilise embankments.

Pond monitoring

The water level in storage ponds should be monitored regularly, along with groundwater heights and sludge depths. To monitor groundwater, a shallow (0–3 m) slotted PVC pipe in which depth is measured is all that is required. Water table heights must not come within 1 m of the compacted lining of the storage, or the compacted clay layer can become damaged, and effluent from the pond can enter groundwater. Monitoring of effluent levels allows any seepage losses to be readily detected and corrected. If water tables become close to levels within storage ponds, synthetic linings must not become damaged, and groundwater must not overtop the pond. Monitor these levels monthly during wet periods, when water tables are likely to rise.

Monitoring sludge depths is critical to ensuring sufficient room in the pond for heavy rain. As sludge accumulates, the capacity for effluent diminishes, so ponds must be regularly desludged (see chapter 2.8 'Desludging and pond closure').

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2.6 Effluent storage requirement

Most of the dairy industry is located in regions with a strong seasonal soil moisture deficit, so the opportunity for effluent to be applied to land must be considered in the design of an effluent storage system. The term ‘storage period’ is used to identify the length of time during which effluent distribution is not appropriate, as indicated by historical climatic data.

Although state authority guidelines should be followed for specific requirements, ‘a generally accepted standard is to design any system to cope with the wettest year in ten’ (ARMCANZ & ANZECC 1996). For ‘environmentally sensitive’ sites (for example, where an overflow will result in pollution of waters or cross property boundaries), use a lower frequency of occurrence or other means of mitigating impacts.

Developing a water budget

Modelling the volume of effluent held in storage (with regular and event-driven inflows), the simultaneous loss via evaporation from a variable pond surface area and evapotranspiration from the reuse area is a relatively complex undertaking, so models such as MEDLI, RUSTIC and ERIM have been developed to help. However, simple spreadsheets using monthly precipitation and evaporation data are appropriate if developed to the following criteria. Indeed, modelling based on monthly data is typically more conservative than daily time-step models (Department of Environment and Conservation NSW 2004) and may offer more robustness and flexibility to system operators.

90th percentile rainfall (and evaporation)

Water budgets to determine storage requirements should be based on the 90th percentile rainfall rather than on mean rainfall. Two approaches to generating the 90th percentile data are valid: the 90th percentile wet year and the adjusted 90th percentile monthly rainfall.

90th percentile wet year: Actual rainfall (and evaporation) recorded during the year where the annual total (YR^{90}) is the wettest in 10 years is used as the basis for design. This is the simplest approach but there may be some irregularities, particularly where heavy rain contributing to the annual total falls in months outside the storage period (e.g. in northern NSW and Queensland with summer-dominant rainfall patterns).

Adjusted 90th percentile monthly rainfall: The 90th percentile rainfall, MR^{90}, is determined for each calendar month and totalled ($\sum MR^{90}$). As the total of individual 90th percentile months is much larger than YR^{90} (the chances of recording 12 consecutive 90th percentile months are low), MR^{90} must therefore be adjusted so that $\sum MR^{90}_{adj}$ equals YR^{90}:

$$MR^{90}{}_{adj} = \frac{YR^{90}}{\sum MR^{90}} \times MR^{90} \qquad (1)$$

This method smooths the monthly rainfall to better reflect the seasonal patterns than does using the 90th percentile ‘wet year’. It may result in a slightly lower annual rainfall than the 90th percentile wet year as, under that method, the year with an annual total ranked one position larger than YR^{90} is used. For example, in Table 1, the adopted 90th percentile wet year (1978) has a total rainfall of 655 mm. By comparison, the 90th percentile annual total (YR^{90}) is 633 mm. In addition, as the ratio of 90th percentile to mean monthly rainfall is greater in summer than in winter, the adjustment tends to

increase the proportion of rainfall received in summer months at the expense of winter (storage period) months.

When using the adjusted 90th percentile month method, use 10th percentile evaporation data (ER^{10}_{adj}).

Table 1. Example calculation of YR^{90} and MR^{90}_{adj} (BOM Station 081125—Shepparton).

Year	J	F	M	A	M	J	J	A	S	O	N	D	Total
1977	40.4	45.9	17.9	37.4	56	66.6	23.1	13.1	35.7	18.7	16.3	5.3	376
1978	**54**	**3.5**	**81.9**	**46.4**	**75.9**	**57.2**	**75**	**43.6**	**67.9**	**29.4**	**80.1**	**39.6**	**655**
1979	31	0.1	5.6	30.2	50.4	25.9	12.5	82.4	88.2	62.9	45.2	1.8	436
1980	25.4	0.1	19.3	73.5	25.5	52	47.6	37.2	26.5	58.5	22.5	51.3	439
1981	57.4	32.2	40.4	5.1	54.2	104.5	96.4	117.4	30.4	12.6	34.4	7.9	593
1982	44.4	1.6	41.7	18.4	27.4	23.1	11.1	5.2	22.7	11.3	4	4.6	216
1983	10.5	3.7	51.9	64.9	82.2	36.5	102.9	62.1	75.9	26.6	31.3	21.5	570
1984	104.7	12.7	33.2	39.5	8.4	12.1	68.8	81.3	37.8	41.9	15.3	1.7	457
1985	4.1	1.1	14.4	27.9	56.9	21.6	21.5	86.6	31.7	61	73.1	93.6	494
1986	1.2	10.4	0	44.2	71.5	18.9	100.9	60.9	58.8	81.3	8	39.3	495
1987	22.4	27.3	12.8	34.8	28.2	79.3	60.9	27.2	31.8	25.4	25.1	32.8	408
1988	39.2	10.7	36.7	62.2	78.1	69	54.6	19.5	63.6	23	75.4	85.1	617
1989	27.5	8.5	123.8	70.9	67.4	63.7	41.6	97.7	24.2	59.8	23.4	22.6	631
1990	7.8	52	11.9	53	38.1	48.3	89.4	67.1	22.3	30.4	9.5	15.2	445
1991	84.4	0.4	6.9	19.9	3.9	144.2	66.9	55.7	69.1	0.2	5.4	46.6	504
1992	8.2	16.7	41.3	21.9	80.8	33.6	33.3	71.1	100.6	109.2	82.7	121.4	721
1993	98.4	40	46.1	2	37	25.2	83.6	46.4	117.4	148.6	32.7	67.8	745
1994	9.7	90.3	33.1	6.1	17.5	58.5	21.7	11.2	22.2	15.4	21.2	4.5	311
1995	80.7	12.8	0.3	41.8	105.9	81.5	126.4	19.2	32.3	67.3	36.2	2.7	607
1996	55.5	48.4	37.1	36.4	4.7	94.2	78.4	39.1	55.5	25.6	25.7	20.6	521
1997	39.9	10	3.1	1.5	55.3	26.2	8.1	51.7	71.2	18	58	4.3	347
1998	15.7	37.7	3.1	71.4	9.4	31.1	54	64.1	47.9	49.1	102.3	4.9	491
1999	4.1	2.6	65.6	45.8	65.8	42.6	33.3	95.6	25.3	28.8	59	65.2	534
2000	8.8	43	24.7	54.2	58.3	43.4	50.8	44.5	50.6	55.6	84.1	6.6	525
2001	48.5	75.7	26.6	25.4	8.6	35.5	31.3	40.8	32.9	77.1	22.4	7.1	432
2002	8.7	51.4	15.3	6.5	17.6	37.5	13.7	11.6	27.7	9	11	4.7	215
2003	30.8	31.8	0.3	84	54.5	50.5	75.2	69.7	27	57.3	23.9	84.4	589
2004	4.3	3.2	3.9	7.3	33.9	45.4	35.1	44.1	74.8	18.6	70.4	40	381
2005	13.7	97.2	2.5	25	2.2	74.9	29.5	80.2	40.5	82.3	59.3	32.9	540
2006	28.7	17.5	3	20	17.2	16.1	32.6	12.6	22.6	0.4	7.6	6.3	185
Mean	34	26	27	36	43	51	53	52	48	44	39	31	483
YR90													633
MR90	81	54	53	71	78	83	97	88	77	81	80	84	929
$MR90_{adj}$	**55**	**37**	**36**	**48**	**53**	**56**	**66**	**60**	**53**	**56**	**55**	**58**	**633**

Note that rainfall is rarely 100% effective(in terms of being available for infiltration by soil), as some is lost via runoff, interception and evaporation or percolation below the root zone. Procedures are available to allow for this loss, using effective rainfall instead (Environment Protection Authority 1991), but have not been used here. Unless site-specific limitations warrant further refinement of the storage requirement, use a conservative volume and actual rainfall to provide the farmer with more flexibility in timing distributions.

Evaporation and evapotranspiration data

Evapotranspiration is the transfer of water from the landscape to the atmosphere, and is a combination of evaporation and plant transpiration. Hydrologists and irrigators often

use measured Class A pan evaporation (E_{pan}) and pan coefficients (k_p) to estimate reference crop (potential) evapotranspiration (ET_o). Alternatively, ET_o can be calculated directly from meteorological data using methods described in FAO (1988).

Both E_{pan} and ET_o data can be provided by the Bureau of Meteorology. If the nearest weather station is not likely to yield representative data for the site, the Queensland Government's Natural Resources and Water website can provide a synthetic dataset for any location in Australia, interpolated from surrounding stations (http://www.nrw.qld.gov.au/silo/datadrill/).

Specific crop evapotranspiration (ET_c) from a disease-free, well fertilised crop grown in large fields under optimum soil water conditions and achieving full production is calculated by using experimentally determined ratios of ET_c/ET_o, called crop coefficients (k_c), where:

$$ET_c = k_c \times ET_o \tag{2}$$

Tables of crop coefficients for common pasture species and crops can be sourced from state departments of primary industries, or more generally from FAO (1988). Crop evapotranspiration under non-standard conditions can be calculated where stresses and environmental constraints may reduce ET_c (FAO (1988).

Irrigation deficit

Any month when the irrigation deficit is less than the minimum depth of application achievable by the irrigation system should be considered part of the 'storage period'. For example, a 6-mm deficit (Table 2) in early spring does not provide an opportunity for reuse, as the minimum depth of application by most pressurised irrigation systems (and certainly flood irrigation) is greater. The storage period for the example shown in Table 2 should therefore be based on a minimum of 4 but preferably 5 months (May to September inclusive). Note that a deficit of 100 mm is equivalent to 1 $ML \cdot ha^{-1}$.

Hydraulic balance

The concept of hydraulic balance can be expressed by the following equation:

$$\text{Change in volume} = V_{effluent} + V_{runoff} + (V_{precipitation} - V_{evaporation}) \tag{3}$$

where $V_{effluent}$ = the volume of effluent determined by the water audit (see chapter 1.2 'Characteristics of effluent and manure')

$V_{runoff} = MR^{90}{}_{adj} \times A_{catchment} \times C_x$

$V_{precipitation} = MR^{90}{}_{adj} \times A_{pond}$

$V_{evaporation} = ER^{10}{}_{adj} \times A_{water\ surface}$

C_x = appropriate runoff coefficient (see next section)

$A_{catchment}$ = area of surfaces generating contaminated runoff

A_{pond} = pond area to centreline of embankment

$A_{water\ surface}$ = surface area from which evaporation occurs.

Because the surface area from which evaporation occurs fluctuates with the volume in storage, it is difficult to model using the spreadsheet approach. For that reason, the liquid surface area at mid-depth is generally assumed as a fixed evaporative surface area. Parker *et al*. (1999) found that evaporation from week-old effluent could be closely approximated by Class A pan evaporation. For any pond with a substantial crust, the evaporation component can be ignored.

2.4 Effluent storage requirement

Table 2. Example water balance (BOM Station 081125—Shepparton).

90th percentile water budget—adjusted monthly															Maximum storage volume
Climate data		J	F	M	A	M	J	J	A	S	O	N	D	Total	
Precipitation	mm	55	37	36	48	53	56	66	60	53	56	55	58	633	
ET_o	mm	229	183	156	90	46	28	32	47	73	111	169	195	1360	
k_c		0.9	0.9	0.9	0.8	0.8	0.7	0.7	0.7	0.8	0.9	0.9	0.9		
ET_c	mm	207	165	140	72	37	20	22	33	58	100	152	176	1182	
Deficit	mm	151	128	104	24	0	0	0	0	6	45	97	118	672	
Effluent volumes															
Volume of effluent	m^3	194	175	194	188	194	188	194	194	188	194	188	194	2281	
Volume of runoff	m^3	75	51	50	66	73	77	90	81	72	76	75	79	864	
Net pond surface flux (precip – evap)	m^3	–149	–137	–104	9	77	106	125	90	41	–1	–74	–100	–117	
Change in storage	m^3	121	89	139	262	343	370	409	365	300	269	188	172	3029	
Reuse & storage requirement															
Irrigation volume required	m^3	9073	7679	6241	1412	0	0	0	0	335	2689	5826	7085	40340	
Distribution event occurs?		*n*	*n*	*n*	y	*n*	*n*	*n*	*n*	*n*	y	*n*	*n*		
Cumulative storage	m^3	481	570	710	0	343	713	1122	1488	1788	0	188	361		1788
Distributed volume	m^3	0	0	0	972	0	0	0	0	0	2057	0	0	3029	
Distribution depth	mm	0	0	0	16	0	0	0	0	0	34	0	0	50	
											Warning				

Assumed data for worked example:
Effluent volume = 6250 L·day^{-1}
Runoff area impervious = 750-m^2 concrete feedpad + 650-m^2 concrete yard = 1400 m^2 total impervious
Runoff area pervious = 3000-m^2 loafing pad
Pond catchment area = 2500 m^2
Evaporation area = 1250 m^2
Reuse area = 6 ha
'Warning' flags a distribution depth exceeding a site-specific limit (in this example, 25 mm·month^{-1}).

Calculation of runoff coefficients

Australian Rainfall & Runoff (IEAust 1987) provides a method for determining an appropriate runoff coefficient. Figure 1 relates the runoff coefficient C_{10} for a 10-year average recurrence interval (ARI) event to the pervious and impervious fractions of the catchment, and to the 10-year ARI 1-h rainfall intensity (${}^{10}I_1$). Where ${}^{10}I_1$ is between 25 and 70 mm·h^{-1}, a line can be interpolated as:

$$C_{10} = 0.9 \times f + C^1_{10}(1 - f) \tag{4}$$

where f = fraction impervious (0.0–1.0)

C^1_{10} = pervious area coefficient:

$$C^1_{10} = 0.1 + 0.0133({}^{10}I_1 - 25) \tag{5}$$

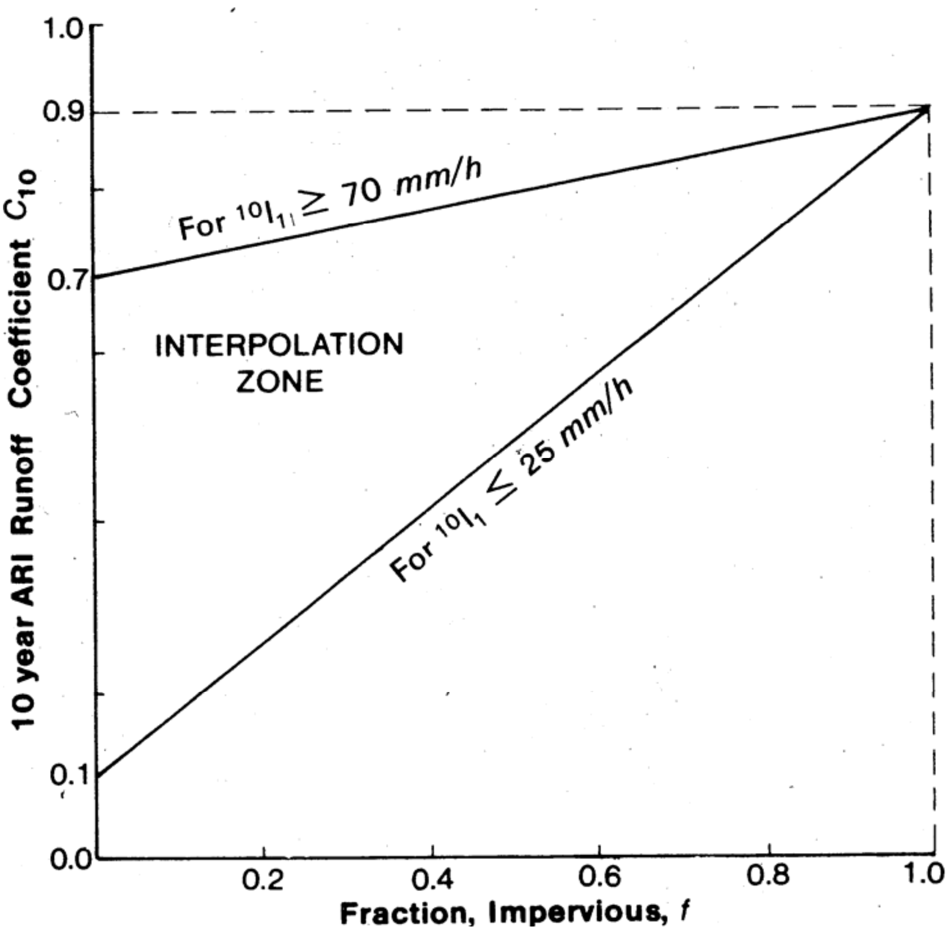

Figure 1. Calculation of runoff coefficient C_{10} (IEAust 1987).

For a balance using 90th percentile rainfall data, it is appropriate to use the 1-year ARI coefficient (C_1):

$$C_1 = 0.8 \times C_{10} \tag{6}$$

Worked example

Data come from Table 2.

2.6 Effluent storage requirement

Effluent volume	6250 L·day^{-1}
Runoff area impervious	750-m^2 concrete feedpad +
	650-m^2 concrete yard = 1400 m^2
Runoff area pervious	3000-m^2 loafing pad
10-year 1-h rainfall intensity $^{10}I_1$	29 mm·h^{-1}
Fraction impervious f	1400 / (1400 + 3000) = 0.32
Pervious area coefficient C^1_{10}	0.1 + 0.0133 × (29 – 25) = 0.15
10-year ARI runoff coefficient C_{10}	0.9 × 0.32 + 0.15 × (1 – 0.32) = 0.39
Annual runoff coefficient C_1	0.8 × 0.38 = 0.31

Maximum cumulative storage volume

The spreadsheet should identify the maximum cumulative storage volume needed to avoid a spill or a distribution event during the storage period. For the worked example in Table 2, the storage volume required was 1788 m^3 (excluding freeboard).

Residual volumes

The preceding section identifies how the 'active' storage volume is determined. In addition to that volume, an allowance needs to be made for any effluent remaining following drawdown and for a treatment volume where a floodwash system uses water from the storage pond.

Allowance for residual volume

The allowance for residual volume should generally be the larger of either:

- a minimum depth of 300 mm to prevent desiccation and cracking of the pond liner; or
- at least 2.5 times the suction inlet (or ~4 times the diameter of the suction pipe) to avoid air entrainment, as the storage pond is likely to be emptied by a pump (APMA 2001). Also see chapter 1.5 'Sump design'.

Storages providing treated effluent for floodwash

Effluent is often recycled from the storage pond to supply a floodwash system for yard and feedpad wash. Sufficient volume should be retained following drawdown to provide a nominal residence time of 20 to 30 days before reuse for aerobic or facultative treatment processes. When applicable, this volume is likely to exceed the volume required for suction submergence or liner protection and becomes the residual volume.

Managing the effluent storage

Depth marker

A properly sized effluent storage is a good start, but it must be managed (drawn down when appropriate) to provide the required storage capacity. A depth marker is a handy device and should be installed with markings showing at least:

- the full supply level (600 mm below top of bank)
- the level at which the 'active' storage volume remains—the water level must be drawn down to this point at the start of the storage period.

It is useful to have additional level marks at increments of 200 to 500 mm to track cumulative volume over the course of the storage period.

Painted level marks can be obscured by discoloration and a build-up of bacterial slime. Use V-shaped notches 25 mm deep instead.

Stormwater diversion and reducing storage requirements

All dairies should:

- collect and use roof runoff by directing gutters to the tank supplying platecooler or washdown water
- reuse platecooler discharge for washdown
- prevent runoff from entering the yards or effluent systems from upslope.

Diverting clean stormwater from the holding yard, however, requires more consideration. Diverting 'clean' runoff from the washed holding yard will reduce the storage volume requirement, pumping costs and the risk of overtopping. On the other hand, farms in dry climates may be required to add fresh water just to dilute pond salinity levels (see Chapter 2.3 'Anaerobic, aerobic and facultative ponds') and would benefit from the addition of clean stormwater. In addition, a storage designed to accommodate stormwater avoids the risk of unintentional diversion of effluent (if the diverter is not reset to 'collect' mode before the yard is washed) and contaminated stormwater. The SA EPA (Environment Protection Authority 2003) suggests that even washed yards may generate contaminated runoff. Therefore, if site constraints allow, capturing stormwater is preferable, and the storage volume should be calculated on that basis. However, farms that practise seasonal milking are usually able to safely divert runoff from cleaned yards over their non-milking periods.

In high-rainfall areas, any or all of the following stormwater minimisation options may be considered:

- Install guttering so that roof runoff is directed to the washdown tank and any overflow is diverted away from the effluent system.
- Reuse platecooler discharge for washdown and divert any excess away from the effluent system.
- Divert runoff from clean holding yards (using a washdown pump cut-out or high-visibility reminder to signal when the diversion is in use).
- Roof the holding yard (which may also help control heat stress in summer).
- Minimise unnecessary accumulation of manure (e.g. by holding cows after milking).
- Install a trafficable solids separation trap to reduce the sludge allowance for the pond and therefore the pond surface area.
- Minimise pond surface area by using a single pond or increasing depth.
- Recycle effluent for yard floodwashing.
- Cover the pond (see chapter 8.1 'Production and beneficial use of methane').

Reuse scheduling

Plan effluent distributions, taking into account the volume in storage, current soil moisture deficits and crop conditions and, where available, 4- to 7-day weather forecasts. See chapter 3.9 'Hydraulic application rate and scheduling'.

Sizing collection and diversion structures for storms

With increasing numbers of uncovered feedpads contributing contaminated runoff from larger catchment areas, it is necessary to consider the rate of runoff that must be dealt with during storms. Intense storms may generate larger volumes of runoff than traditional collection pipes and pumps can handle, causing effluent to overflow the collection system.

National guidelines for beef feedlots require that collection and diversion structures (diversion banks, catch drains, sedimentation basins etc.) be designed to carry the peak flow rate resulting from a design storm event with an ARI of 20 years (ARMCANZ 1997). The design storm has a duration equal to the time of concentration, that is, the time taken for runoff to travel from the most remote point in the catchment to the point of interest (IEAust 1987) both by overland flow and in any drain or pipe.

Owing to the limited extent of most collected surfaces around a dairy, it may be preferable in all but the largest operations to adopt a conservative time of concentration; that is, a time of concentration of 5 min will give the highest-intensity rainfall event. Check the capacity of all structures to ensure that they can handle the expected peak flows.

The rate of runoff (Q, $m^3{\cdot}s^{-1}$) is given by:

$$Q = \frac{C\,i\,A}{3.6x10^6} \qquad (7)$$

where C = runoff coefficient (see ‘Developing a water budget’ above); $C_{20} = 1.05 \times C_{10}$

i = intensity ($mm{\cdot}h^{-1}$)

A = catchment area (m^2).

Local storm intensities can be obtained from the Bureau of Meteorology (http://www.bom.gov.au/hydro/has/ifd.shtml).

Worked example

A 150-mm pipe is proposed for both the effluent collection pipe and stormwater diversion device for the 650-m^2 yard (above). The 5-min storm intensity in this worked example was determined to be 125 $mm{\cdot}h^{-1}$ (IEAust 1987).

Fraction impervious f	= 1.0
10-year ARI runoff coefficient, C_{10}	= 0.9 × 1.0 = 0.9
20-year ARI runoff coefficient, C_{20}	= 1.05 × 0.9 = 0.95
Runoff, Q	= (0.95 × 125 × 650) / 3 600 000
	= 0.02 $m^3{\cdot}s^{-1}$ (20 $L{\cdot}s^{-1}$).

At a grade of 1 in 60, a 150-mm (ID) pipe will convey approximately 30 $L{\cdot}s^{-1}$. Therefore, the proposal is adequate.

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2.7 Wetlands

Both natural and constructed wetlands have been used to capture farm runoff and, in rare cases, to treat dairy effluent.

While it is possible to use constructed wetlands as one component of a wastewater treatment system, their value for treating dairy effluent has not been demonstrated. The use of wetlands linked to waterways, usually practised in districts where wetlands are perennial rather than ephemeral, is strongly discouraged.

Natural wetlands

Natural wetlands have dwindled markedly since European settlement. Despite regulations on the disposal of effluent, the discharge of farm runoff and dairy effluent has helped maintain some natural wetlands. But the associated changes in nutrient, salt and water balances are rarely positive.

The range of natural wetlands which have been modified to receive effluent is difficult to quantify owing to landscape modification and loss of habitat. Some sites are acceptable owing to their isolation and impervious soil, but impounding wastewater in permeable stream beds cannot be condoned. Downstream water quality has no doubt received a measure of protection from some sites, but rarely can these sites be managed effectively to demonstrate isolation from the catchments in which they are installed. Some wetlands holding dairy effluent are dewatered by pumping during times of low flow time and allow passage of flood flows. This use is expedient rather than optimal and requires extensive long time with minimal human influence. Upon effluent discharge, most of the area may monitoring to demonstrate compliance with regulations.

The hydrology and associated hydraulic regime in natural wetlands have evolved over a be ‘wetted’, but owing to channelisation, most of the water flows through a relatively small proportion of the total wetland. Only a small volume of the effluent may come into contact with parts of the wetlands which offer the best treatment prospects. It is not possible to correct this problem by limited land forming and installing banks while preserving the values of the original natural wetland. The lack of control and the presence of an open system limits the value of natural wetlands for dealing with dairy effluent.

Declining water quality in domestic water supply catchments and blue green algal blooms in the early 1990s placed controls on wetland exploitation. The onset of drought in the late 1990s favoured the recycling of effluent and emphasised the need to use constructed wetlands for treatment to avoid the further deterioration of existing waterways and associated wetlands.

Rarely are natural or modified wetlands subject to design criteria for their use as effluent treatment facilities. Monitoring indicates that the characteristics of influent entering natural wetlands vary markedly, ranging from BOD levels of $<10\ mg{\cdot}L^{-1}$ to $>500\ mg{\cdot}L^{-1}$. Despite this range, the quality of the treated effluent is usually fairly consistent and usually <10 mg BOD L^{-1}. The introduction of sedimentation traps before entry and the installation of a conventional treatment pond to slow down the rate of influent flow bring dramatic improvements in the performance of natural wetlands.

Constructed wetlands

Constructing a wastewater treatment wetland in a terrestrial landscape where no wetland existed before avoids the regulatory and environmental entanglements associated with natural wetlands and allows for the design of the wetland for optimum hydrological performance, hydraulic flows and enhanced wastewater treatment. A

constructed wetland should yield higher-quality effluent than a natural wetland of equal area, since the inflow and outflow can be regulated, the bed can be graded and the period of detention can be controlled. Process reliability is also improved because the vegetation and other system components can be managed as required. Of primary importance is the recognition that a constructed wetland serves a role in a waste management system and is not isolated. Research confirms that the performance of a wetland as a treatment stage is contingent upon the nature of detention and the period of wastewater storage before receipt of additional wastewater. If the previous stage of treatment is inadequate, the wetland will not compensate for poor performance.

A range of propriety wetland systems are used for polishing wastewater following primary and secondary treatment. The design criteria for a range of facilities are provided in Hammer (1989), Kadlec and Knight (1996), Polprasert (1996) and Reed *et al*. (1995). Additional research on constructed wetlands for dairy effluent was undertaken at Ruakura, New Zealand, in the mid 1990s (MAF 1997) and in the Hunter region of NSW (DRDC 1997). Although these studies demonstrated that effluent quality improved, they failed to demonstrate the economic viability of the systems under study.

Wetland treatment systems are generally divided between free-water-surface and subsurface-flow systems. In a free-water-surface wetland the water surface is exposed to the atmosphere, and the bed contains emergent plants, soil for rooting, a liner to protect the groundwater, and inlet and outlet structures designed to distribute wastewater evenly. The wastewater depth ranges from 20 mm to 800 mm or more, depending on the purpose of the wetland (batch or continuous treatment) and on whether or not it can be permitted to dry out. A normal operating depth of about 300 mm is required for the maintenance of aerobic conditions through the penetration of sunlight. Most of the wetlands evaluated for treating dairy effluent were of this type.

A subsurface-flow wetland consists of a basin filled with a porous medium, usually gravel, in which the water level is maintained below the top of the gravel. The depth of the gravel is typically 300 to 600 mm. The vegetation is planted in the upper part of the gravel. The same plant species are used as in free-water-surface wetlands, with the exception of floating macrophytes. A liner may be needed to protect groundwater.

There is no single design criterion for sizing constructed wetlands; the techniques used include:

- multiple regression analysis of performance data from operating systems to derive design criteria that can then be used as a ‘recipe’—the experimental ‘suck it and see’ approach
- an areal loading approach in which performance is related to the volume of effluent or mass of organic matter entering per unit time divided by the surface area; this assumes that the wetland behaves like an aerobic pond
- a biological reaction approach that assumes that the wetland responds to waste in a similar manner to other attached-growth-media treatment systems; this assumes minimal particulate matter in the influent.

Organic matter loading rate

Most wetlands can cope with daily organic loads of up to 100 $kg{\cdot}ha^{-1}$, but with higher loadings it is recommended that proprietary design loadings be followed. Hydraulic residence time is a major factor: a minimum period of 3 days is specified for subsurface flow systems. Under this requirement, the surface area and storage requirements for wetlands catering for dairy effluent are very high. The free-water-surface wetland demands a large surface area, yielding high evaporation losses and salt concentration, but it is generally cheaper and easier to construct and maintain than a subsurface flow wetland.

There are advantages and disadvantages with both systems. The biological reactions in both types of wetland are due to attached growth organisms. Since the gravel medium

has more surface area than the open-water-surface wetland, the gravel bed will have higher reaction rates and therefore can have a smaller area. Since the water surface is below the top of the medium and not exposed, the subsurface-flow type does not have the mosquito problems associated with an exposed water surface and suffers from lower evaporation rates. Much greater experience has been derived from research on free-water-surface wetlands serving dairy farms than from subsurface wetlands. The latter type is used mainly for urban applications, individual households and polishing secondary-treated effluent before waterway discharge.

Both types of wetlands rely upon the formation of a biofilm to provide contact between nutrients in the wastewater and organisms in the wetlands. The film is made up of a consortium of bacteria, fungi and algae embedded in a polysaccharide matrix. It provides a critical mass of microbes which absorb and retain organic and inorganic colloids and nutrients, and produce enzymes that act on particulate and dissolved organic material. It forms a potential external energy reserve for low light situations, night time or when there are stark changes in organic loading.

The rate of breakdown of molecules by hydrolytic enzymes determines the rate of decomposition of organic materials. In this process, large organic molecules are broken down to a size which bacteria are capable of assimilating. Aquatic plants continually supply organic material to the microbial layers in their root zone. This supply maintains the concentrations of enzymes that hydrolyse polymeric material in the nearby biofilm. In exposed sediment, the enzyme concentrations in biofilms are much lower, supporting the use of media with a high surface area to volume ratio. The plants are important also because they transfer oxygen to the sediment via their roots, maintaining aerobic conditions for wastewater treatment.

Performance of wetlands

The rates of removal of settleable organics in well designed wetlands are very high on account of quiescent conditions in free-water-surface systems and deposition and filtration in subsurface-flow systems. BOD removal rates vary, but usually range from 50% to more than 90%. If both macrophytes and microphytes (such as algae) are harvested, good nutrient removal rates can be achieved; if not, nutrients accumulate and only N is exhausted. P and K concentrations in treated effluent are reduced, but the medium traps the surplus, increasing the concentrations.

All wetlands rely on macrophyte growth, and anything which compromises this growth detracts from the treatment process. Wetlands need to be well designed in terms of hydrology and hydraulics. If the flow rate is too great and contact time between the effluent and the medium is not adequate, treatment will deteriorate. If the flow rate declines and the wetland receives excessive solids, the contact area will be reduced and the wetland could clog. Similarly, if the macrophytes and algae grow excessively, the root and algal mats can clog flow passages as they die. Although wetlands are effective treatment systems for removing suspended sediment and reducing BOD, pathogens and N loads, they accumulate P, K and trace elements when the plants are not harvested.

For dairy farm use, wetlands need a lot of relatively flat land and need to be managed. Because it is difficult to harvest the plants, wetlands fail to make effective use of nutrients on a farm and encourage loss of water through evaporation and transpiration from plants of lower economic significance than crops or pastures. Their main application appears to be in effluent polishing and for improving the quality of farm runoff, which generally has a high volume and a high organic matter content.

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2.7 Wetlands

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2.8 Desludging and pond closure

The reduction in concentrations of solids, BOD and particle-bound nutrients in ponds is due largely to sedimentation (see chapter 2.3 ‘Anaerobic, aerobic and facultative ponds’). Sedimentation can be problematic in ponds, as the accumulated sludge reduces the active pond volume and residence time and, consequently, treatment efficiency.

The choice of desludging option is an important consideration during the design stage. Options for desludging usually require batch removal:

- by excavator (standard or long-reach) or dragline with subsequent dewatering
- by vacuum tanker, with or without prior agitation
- by sludge agitation and pumping to a tanker, a big-gun irrigator or a drag-hose injector.

More frequent solids removal is offered by *in situ* sludge removal pipes and semi-continuous pumping to a dewatering bay.

Sludge measurement and characteristics

Sludge is a black, gritty, tar-like material that comprises a mix of inorganic material (sand etc.), slowly digestible organic material and dead microbial cell mass. It has a small particle size and is not readily separable in a solid–liquid separator. The sludge layer is a mobile fluid that forms peaks and valleys within the pond.

The basic principles of sludge management are adapted from Sheffield *et al.* (2000):

- Minimise sludge accumulation where possible by using a solids trap.
- Identify the trigger point for sludge removal—i.e. identify the depth where the volume of sludge begins to reduce required active volume (see chapter 2.3 ‘Anaerobic, aerobic and facultative ponds’). Record the trigger point permanently on a depth marker (for single ponds).
- Monitor sludge build-up. In a single pond system, the water level is regularly drawn down during irrigation, exposing the trigger level on the depth marker (see chapter 2.6 ‘Effluent storage requirement’). In a multiple pond system, water level in the primary (anaerobic) pond usually does not fluctuate, so take direct measurements (see below) before the end of the anticipated clean-out period.
- After the trigger point is reached, remove sludge (see ‘Sludge removal’ below), but leave a small residue (~150 mm or so) in the base to re-seed microbial activity upon refilling and to prevent the liner from drying out. Protecting the integrity of the liner during sludge removal is critical—aggressive agitation or over-enthusiastic excavator use may damage the liner and contaminate groundwater.
- Reuse nutrients via land application at agronomic rates (see chapter 3.1 ‘Nutrient budgeting’).

The depth of the sludge is typically measured by probing from a boat (a slow, inaccurate and potentially dangerous task). A lightweight pole, sometimes fitted with a bottom plate, is lowered slowly into the lagoon until the liquid becomes denser; at that point the depth is recorded. The pole is then pushed lower until the bottom of the lagoon is reached, and the depth is again recorded. The difference between the two readings is the sludge depth. At least 10 depth measurements, including one at the marker, need to be made in a representative survey (Sheffield *et al.* 2000). Avoid

locations with pipe inlets or pump intakes, where the sludge level is likely to be disturbed by localised flow patterns.

It is often very difficult to determine the interface between liquid and sludge, particularly in deeper ponds. Researchers sometimes use a nephelometer (light reflectance meter) to improve measurement accuracy. More recently, sonar has been used. The Queensland DPI (Duperouzel nd.) found that sonar in piggery effluent ponds offers rapid sludge measurement with an accuracy comparable to the nephelometer. Singh *et al*. (2007) reports on the development of a prototype that can map sludge profiles without requiring a person in a boat. At this stage, only a few commercial contractors can provide a sonar service.

Sludge samples for testing can be taken from a boat before agitation by using a length of 18-mm PVC tube:

- While wearing gloves, insert the tube to the base of the pond.
- Place your thumb over the open end and slowly withdraw the tube while maintaining the vacuum.
- Hold the lower end of the tube over a bucket and release your thumb, collecting only the black sludge.
- Repeat until 8 to 10 samples have been collected.
- Mix the samples and extract a subsample for analysis.

Procedures for sampling solids from a dewatering bay are given in Redding (2003). Also see chapter 7 ‘Monitoring and sampling’.

Typical characteristics of sludge are shown in Table 1. Note, however, that after surveying 30 piggery effluent lagoons, Sheffield (2000) concluded that owing to the variation in nutrient concentrations in sludge between farms, an analysis of the sludge in question was necessary for calculating application rates rather than using ‘typical’ values.

Table 1. Published sludge characteristics (standard deviation in parentheses).

Parameter	Units	Longhurst *et al.* (2000)	Barker *et al.* (2001)	pers. comm. G. Ward, 2007, QDPI.
Density	$kg{\cdot}m^{-3}$		994	
TS	%		7.3 (4.6)	6.5–8 [a]
VS	% (of TS)		57 (5.9)	
COD	$mg{\cdot}L^{-1}$		31206 (19002)	
pH			7.5 (0.55)	
Total N	$mg{\cdot}L^{-1}$	2450		1000–1400
TKN	$mg{\cdot}L^{-1}$		2276 (1042)	
NH_4-N	$mg{\cdot}L^{-1}$			210
NH_3-N	% TKN		32 (23)	55
P	$mg{\cdot}L^{-1}$	250	2197 (1726)	190–192
K	$mg{\cdot}L^{-1}$	500	918 (719)	620–625
S	$mg{\cdot}L^{-1}$			370
Na	$mg{\cdot}L^{-1}$		347 (192)	75
Ca	$mg{\cdot}L^{-1}$			2174
Mg	$mg{\cdot}L^{-1}$			872
Cu	$mg{\cdot}L^{-1}$		55 (44)	
Zn	$mg{\cdot}L^{-1}$		89 (49)	
EC	$\mu S{\cdot}cm^{-1}$		3649 (726)	

a: Generally, effluent with <8% solids can be pumped. The consistency of the sludge recovered by Ward (2007) was limited by the capacity of the vacuum tanker to extract the settled material without agitation.

US sources suggest that sludge has a lower N:P ratio than effluent, so application rates must be adjusted accordingly (see chapter 3.1 'Nutrient budgeting'). However, Australian (Ward 2007) and NZ (Longhurst *et al.* 2000) data show N:P ratios not dissimilar to those of effluent.

Sludge removal

Removal for reuse as a solid

- Pump liquid above sludge by usual distribution system.
- Using an excavator and mud bucket, remove sludge and place, via a sealed truck, into a bunded dewatering bay or one that drains back into the pond.
- After air-drying, solids can be hauled and spread with conventional solids-handling equipment.

Do not dump wet sludge directly onto paddocks: extremely high nutrient loading rates and the risk of leachate contaminating surface waters or groundwaters preclude this.

Removal for reuse as a slurry

- Pump liquid above sludge by usual distribution system, leaving sufficient behind to dilute sludge to a pumpable state following agitation.
- Agitate remaining contents and remove via a vacuum tanker (equipped with surface spray plates or soil injectors), or pump to a slurry spreader, big-gun irrigator or umbilical hose injector.

Agitation equipment can be either PTO-powered propeller-type mixers or hydraulic agitator/pumps that can also load slurry spreaders. Both types of agitators have a limited radius (~15 m; Jones *et al*. (2006)), so access points are required at least every 30 m.

Agitators can erode earthen liners, so propeller types must be kept at least 1 m from the liner. Hydraulic agitators must be monitored to ensure that the recirculation jet is not scouring the embankment. Check periodically for leaks from hoses and couplings to avoid spills and impairing vehicle traction or access.

Some contractors prefer to at least one of the long sides of the pond to be 6 m wide to allow for machinery access during desludging. Earthen ramps with a grade of 1:10 will allow safe approach to and departure from the embankment. It is also beneficial to provide a gravel-topped crest to maintain good traction while machinery is working beside the pond. Remember that such machinery can weigh in excess of 30 tonnes, and OH&S issues must be considered during the design.

In situ sludge removal—emerging technology

Some covered anaerobic ponds are equipped with a network of pipes across the base to:

- remove sludge without removal of the cover
- inoculate influent with microbes from the recovered sludge to enhance biogas production (see chapter 8.1 'Production and beneficial use of methane').

Such sludge harvesting techniques are also potentially useful for uncovered ponds, as they offer:

- the opportunity to minimise the required pond volume via a reduction in the sludge allowance (lower construction costs, lower total odour emissions)

- regular sludge removal with greater retention of nutrients for reuse
- avoidance of disruptive batch desludging.

In covered anaerobic ponds, a pump capable of handling high-solids material (up to 8%) is connected to the pipe network via a multi-valved manifold and operates semi-continuously, switching between laterals to remove sludge and return it to a mixing tank or solid–liquid separation system.

Typically the design of any such system is proprietary knowledge. However, the basic principles of pumping sludge require the following features:

- A minimum scour velocity of 1 $m \cdot s^{-1}$ (see chapter 1.6 'Pipes'), but typically not greater than 2 $m \cdot s^{-1}$, depending on pump characteristics (net positive suction head required).
- A pump that can handle solids and abrasive material (sand is more of a problem in dairy effluent than in piggery effluent) yet has reasonably good efficiency to minimise life-cycle operating costs.
- A compromise between the number of inlets and the numbers of laterals and valves to minimise installation costs without limiting the removal of sludge by 'rat-holing'. Rat-holing is the situation where one inlet on a lateral with many inlets may clear faster than the others, allowing effluent into the dewatering bay. Installing more laterals with one inlet each offers improved control, but the sludge's *in situ* angle of repose and therefore inlet spacing have not yet been documented.

The adoption of such systems is likely to be possible only where a drying bay is practical; that is: there is a sufficient area of flat or gently sloping land; the *in situ* soils allow construction of a pad with an impermeable clay liner; and the climate includes drying periods to allow evaporation to reduce the sludge moisture content from the 'slurry' range to the 'solid' range.

Sludge dewatering

Sludge dewatering bays

A sludge drying bay or pad may be simple or elaborate depending on the intended frequency of use. It has the following functional requirements:

- The pad must be able to drain any leachate or contaminated runoff back into the effluent collection system, or have a bunded volume that contains all leachate and any runoff.
- The base must be relatively impermeable and should be prepared with a compacted earthen liner (or have an artificial liner and a means of protection). Skerman (2005) provides a list of recommended construction procedures for earthen pads.
- Some form of bunding is required, as the moisture content of the removed sludge is too high for it to be stackable. This may be provided by earthen embankments, large bales of straw, or geotextile or shadecloth fences.

2.8 Desludging and pond closure

Figure 1. Empty drying bay with shadecloth fence (photo courtesy of Australian Pork Limited & QAF Meats).

Figure 2. Drying bay before clean-out (photo courtesy of Australian Pork Limited & QAF Meats).

A geotextile or shadecloth fence around a drying bay offers higher dewatering rates by virtue of its permeability. A simple shadecloth fence dewatering bay is shown in Figure 1. Note that the layer of sand on the pad indicates the base during clean-out.

Structures similar to separation and evaporation ponds (see chapter 2.1 'Solid–liquid separation systems') may be used to minimise costs. Design depths of approximately 500 mm for wet sludge will promote faster drying during the drier months.

Geotubes

Geotubes have also been used to dewater high-solids-content effluents and pond sludge. A large geotextile fabric tube is placed on an impermeable pad, filled with effluent, and then left to dewater for 2 to 5 days. After the tube dewaters sufficiently, additional effluent can be pumped in to refill it. This process of filling and dewatering may be repeated several times until the tube is full of solids. A final dewatering period of 10 to 14 days (or longer) can be used before the tube is opened and the solids are removed. Once opened, the tubes are discarded (or recycled for other uses such as laneway stabilisation or weed mats).

The tube retains a high percentage of the solids (95% TS, 80% TKN, 80% TP, 30% K), and the liquid returns to the effluent system. The solids can remain in the tube until spreading and do not need to be covered (rainfall will not enter the tube).

Chemical coagulants and flocculants can be added to the effluent to enhance nutrient removal and to speed the rate of liquid drainage from the geotube so as to shorten the time between refills (see chapter 5 'Odour emissions and control').

When used in pond desludging operations, the geotube system is limited by being batch-loaded: dewatering requires time during which specialist sludge agitation and pumping equipment (usually hired) sits idle. As sludge pumping costs may exceed the cost of the bag itself, ponds should be drawn down (by the usual irrigation procedures) to the sludge level before pumping sludge to maximise the solids content and reduce the number of refills. Two or more tubes may be warranted to maximise the use of specialist pumping equipment.

It is conceivable that a relatively large pair of geotubes (or more) could be used to separate solids continuously. However, the continuous drainage of effluent would probably necessitate construction of a concrete pad with drainage collection under the area. If earthen pads constructed with a compacted clay or membrane liner were used, bunding between tubes would be necessary to maintain access by isolating drainage from the working tube, and the liner would need to be protected from damage during tube removal and replacement.

The cost of a 14-m × 30.5-m geotube in 2005 was approximately US$10 m^{-3} (Texas Water Resources Institute nd.).

Pond closure

When a dairy ceases operation or a pond is to be replaced, existing ponds need to be closed properly so that they do not constitute a risk to surface water or groundwaters. Jones *et al.* (2006) suggest the following options as appropriate strategies for the permanent closure of ponds.

Option 1—Permanent elimination of earthen storage structure

1. Divert all surface water runoff away from the storage.
2. Remove any pipes and structures adding effluent to the storage.
3. Remove all liquid, pumpable sludge and solids.

4. Fill the structure with soil by pushing in existing embankments and bringing in additional fill as needed. The degree of compaction required for backfill material will depend on the anticipated future use of the site, but must be sufficient that settlement does not create a depression that collects rainwater. The backfill height should exceed the design finished grade by 5% to allow for settlement.

5. Establish a crop cover to minimise soil erosion. A crop with deep roots such as lucerne is preferred because of its ability to draw up any remaining nutrients.

Option 2—Permanent conversion to a freshwater pond

1. Add an overflow spillway (if one does not exist) or a standpipe to set a maximum water level at least 0.6 m below the lowest point in the embankment.

2. Remove any pipes and structures adding runoff or effluent to the storage.

3. Remove all liquid, pumpable sludge and solids.

4. Immediately after clean-out, refill the pond with fresh water to prevent the liner from drying out. When conditions suit irrigation, agitate the pond and completely empty it.

5. Refill the pond with water. If the resulting water quality meets the objectives for agricultural irrigation water given by ANZECC & ARMCANZ (2000), the structure can be managed as a farm pond. Otherwise, continue the cycle of emptying and refilling.

Note that regulatory controls on the establishment of farm dams may apply.

Option 3—Breaching the embankment

1. Divert all surface water runoff away from the storage.

2. Remove any pipes and structures adding runoff or effluent to the storage.

3. Remove all liquid, pumpable sludge and solids.

4. Breach the embankment low enough to allow any water that enters the pond to quickly drain away (this option may not be possible for below-ground structures).

5. Establish a growing crop or pasture. A crop with deep roots such as lucerne is preferred because of its ability to draw up any remaining nutrients.

Breaching the embankment before all contents are removed is not recommended: pollution incidents (and prosecution) have resulted from such an approach.

Option 4—Managing storages on temporarily de-stocked farms

Where a farm is temporarily de-stocked with the intent to restart later, a fourth option is appropriate.

1. Divert all surface water runoff away from the storage.

2. Remove all liquid, pumpable sludge and solids.

3. Refill the storage with water to limit damage to the liner from desiccation, weed growth, erosion and burrowing animals.

4. Manage the storage to prevent liquid overflow.

Under any of these options, it is important to protect the existing earthen (or geotextile) liner from damage. An intact liner minimises the risk of pollution of groundwater.

The cost of closing a pond is significant, and a lack of funds upon ceasing operations may be a deterrent to following the recommended procedure. However, if an improperly closed pond causes pollution, the costs in mitigating the damage and possible fines would exceed any avoided costs.

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2.9 Composting

The management of manure by composting does not simply consist of placing the accumulated manure in a pile and leaving it to rot down. The process relies upon control of temperature, moisture and feedstock and on supplementation of the manure with straw, sawdust or hay to serve as a source of carbon. Information on standards for composting for soil conditioners and mulches is provided in Standards Australia (2003). Further information is presented in Recycled Organics Unit (2007). The Recycled Organics Unit website is particularly valuable if the compost is to be marketed.

The composting process

Composting is the breakdown of relatively dry manure by microorganisms and fungi under aerobic, moist conditions. The naturally elevated temperatures foster microbial growth, kill weed seeds, encourage pathogen die off, kill helminths and cysts, and avoid generation of noxious gases. During composting, the readily biodegradable component of the waste is oxidised (converted to carbon dioxide, water and heat), leaving an organic residue (humus).

The metabolic heat generated by the microorganisms elevates the temperature of the mixture. The heat, if not too high (<60 °C), promotes rapid decomposition as a result of the build-up of microbial biomass. Temperatures in excess of 55 °C for 3 days are effective in killing weed seeds.

Success with composting depends on providing conditions conducive to the preferential growth of desirable microbes. It is not an *ad hoc* process but needs careful management for success; viable large-scale operations are uncommon. Anaerobic conditions and noxious odours are common problems, particularly at the larger scale. In addition, contaminants in the manure such as antibiotics and disinfectants, and changes in pH, moisture content, temperature and feedstock, can hamper microbial activity.

Generally the main objective of composting is to increase the nutrient density and nutrient availability of manure with minimal mechanical processing and odour via the control of a biological process. This process assists storage, transport and reuse. The advantages and disadvantages of composting and related manure treatment processes are discussed in Pittaway *et al*. (2001).

Advantages of composting

- Composting and sale of the product will be more cost effective and environmentally friendly than some other management options, including storage and landfill.
- Composting can have less impact on the environment than most alternatives. Current research aims to reduce greenhouse gas emissions.
- A biologically stable compost does not generate noxious odours during land application and can be stored without being a nuisance because it forms a water repellent crust.
- Stable compost does not provide a medium for the breeding of flies.
- Unlike some organic wastes (including sludge, barley, sawdust, green waste and food processing waste), mature manure compost does not contain or produce phytotoxic substances (which inhibit plant growth and seed germination).
- The heat generated during composting promotes moisture removal, with the result that it is less costly to store and transport the composted material than the raw manure.

- Heat and other factors generated during composting destroy pathogens and most common weed seeds.
- Plant nutrients in the organic material become more concentrated as the readily biodegradable carbon compounds are removed and the manure volume is reduced.
- Compost contains both macro- and micronutrients.
- Compost has a higher nutrient density and availability than raw manure, thereby improving the cost-effectiveness of reuse.
- The structure and appearance of the organic material is improved, making it easier to reuse or sell.
- The product is more homogeneous, i.e. less lumpy and easier to handle and spread.
- There is likely to be more community support for the transport and application of compost than of manure.
- The manure to be composted can be mixed with other sources of waste organic material such as garden refuse, sawdust, straw, food scraps and forest prunings to provide a value-added resource.
- Well managed compost generates less greenhouse gas than stockpiled manure.
- The result is a higher-value product suitable for use in high-return industries such as horticulture and urban landscaping.

Disadvantages of composting

- The effectiveness of the composting operation is usually dictated by atmospheric conditions, and quality suffers during wet, cold or dry weather.
- A high degree of control of moisture and temperature is required to achieve a satisfactory product.
- The material must have a relatively high void ratio, warranting the use of low-density blending agents.
- The markets for compost are not as well defined as those for commercial fertiliser or animal manure, and the characteristics of the bulking agent can affect the quality of the compost.
- High application rates are required to meet crop nutrient requirements.
- Cartage costs are higher than for fertilisers.
- Composting is more demanding than the direct application of manure to land.
- The high capital and operating costs of the required turning machinery.
- Labour costs extra.
- There is a risk of odour generation during the composting process.

Composting methods

Methods of composting vary. Some proprietary systems require licences to operate. The windrow system, with little process control and a long stabilisation period, is common. The totally enclosed composting reactor, with a high capital cost, complex design and high level of process control, is not commonly used for dairy manure. Dairy manure can, however, be added to regional facilities (if available) to enhance the process. The selection of the type of composting method depends on the scale of the enterprise, age and source of the manure, site and landscape characteristics, climate,

proximity of neighbours, availability of expertise, funding and a source of cheap blending agent.

Windrows

Windrow composting relies on turning and passive aeration, whereas forced aeration composting uses mechanical systems for process control. The suitability of each method is dictated by waste characteristics such as porosity and content of readily biodegradable material, and by site factors, including proximity of neighbours, access to carbon sources (blending agents) and power availability. Windrow composting is favoured for manure processed in a rural setting. Forced aeration is used to yield-higher quality compost for application in urban and urban-fringe situations, particularly at nurseries and in mushroom culture. Apart from simple stockpiling, the windrow method is the most common method used for stabilising dairy manure with bulking agents or carbon sources such as straw, sawdust or rice hulls. Two types of windrow are used: one relies on static stockpiles, the other on turned stockpiles; both can be watered. Temporary covers can be used to preserve moisture or shroud stockpiles from rain..

Static windrows

The static windrow method relies on passive aeration to provide oxygen. Oxygen enters the pile through a combination of diffusion and convection (caused by heating within the pile). Moisture can be added by sprinklers or spray carts.

The aspect and size of the windrow and the porosity of the material affect how well passive aeration works; these factors also affect heat loss, and thus the internal pile temperatures. Stabilisation is enhanced by controlling the size and porosity of the windrow so that it is both small enough (in cross-sectional area) and ‘fluffed up’ enough to allow adequate oxygen transfer, yet large enough by critical mass to retain some heat.

Static composting is suitable only for manure with a low content of readily biodegradable substrate and an open structure achieved by mixing with straw, sawdust, rice hulls or leaves. Manure with greater oxygen demand may be composted by this technique when diluted with coarse, porous, inert blending material. The addition of a blending (or bulking) material also produces an open structure in the mixture. Anaerobic conditions and suboptimal temperatures and moisture levels are unavoidable with static stockpiles. Accordingly, stabilisation periods may range from 6 months to years depending on prevailing atmospheric conditions. Siting of static windrows is critical to facilitating airflow and to taking advantage of sunlight and exposure. Shade should be avoided. To ensure the homogeneity of product and uniformity of process time, the static stockpiles should be subject to the same conditions of exposure; this usually dictates a north–south alignment.

Turned windrows

Turned windrows are similar to static windrows, except the material is turned or agitated to introduce air into the stockpile and bulk it to facilitate homogeneity, passive aeration and heat removal.

A front-end loader or specialised turning machine is commonly used. With turned windrows, fresh manure with a higher oxygen demand can be composted more rapidly, and larger windrows can be used than with static windrows. However, mechanical turning cannot control compost temperatures precisely, and unless the material is turned frequently, anaerobic conditions are unavoidable. Water is often added to promote effective biological activity in the summer, via either overhead irrigation or a water cart and side sprinkler system.

The frequency of turning required to prevent nuisance odours and achieve rapid stabilisation will depend on the manure's oxygen demand and porosity and on the type and amount of blending agent and carbon source. Stabilisation periods of less than 3 months are achievable. Avoid areas prone to katabatic wind drift, as odour generated under cooler temperatures during the night can flow downhill. Labour and site access can be limiting, and neighbours can encounter odour.

Management and quality control

Given appropriate initial feedstock, the most important factors influencing the rate and efficiency of composting are oxygen supply, temperature control and availability of water in the blend. Oxygen is required by the composting microorganisms to oxidise biodegradable material. The higher the content of readily biodegradable material, the greater the potential oxygen demand. If insufficient oxygen is supplied, anaerobic conditions will result, reducing quality, producing noxious gases and generating more greenhouse gases. The aim of composting should be to yield consistent product quality. To achieve this, the process must be monitored. In particular, the finished product should be analysed for pH, moisture and nitrogen, especially if it is to be sold.

Bulking and blending agents and carbon sources

The structure and moisture content of a solid waste determines how easily it can be aerated, and on the type and quantity of blending and bulking agents or carbon sources required (if any) to enhance porosity and absorb excess moisture. The method of mixing the manure and bulking agent will also be determined to some extent by the structure of the manure. Blending and bulking agents must maintain their structural integrity during the composting process to provide air voids. Commonly used agents and sources of carbon for dairy manure are:

- rice hulls
- food processing by-products (orange peel, pips, husks)
- woodchips
- sawdust
- crushed pine bark or other bark with a granular structure when crushed
- organic wastes with structural integrity such as leaves, grass and straw
- recycled compost
- dried dairy manure or sludge.

Recycled compost can be used as a bulking agent for compost, although Pecchia *et al*. (2002) indicate that there is little benefit in this practice.

Approach local industries which produce waste material suitable for use as bulking agents for access to feedstock. Blending and bulking agents can be used in combination, and if coarse bulking agents such as woodchips are used, the bulking agent can be screened from the compost and reused. Degradable agents are generally preferred to those which simply provide structure to a pile.

Pittaway *et al.* (2001) discuss regional solutions to waste disposal by composting. This type of solution is favoured if the amount of material generated from farms is significant enough to justify investment. Urban garden waste is frequently used as the main carbon source, and operations are now common on the fringes of many cities. Factors other than the structure of the waste that affect the selection of a blending or bulking agent are:

- availability and cost

- product quality requirements (e.g. crushed pine bark normally produces a compost with a better appearance than do sawdust or rice hulls)
- product volume constraints; recycling the compost or bulking agent (after separation by screening) reduces the volume of product
- consistency of supply
- colour
- risk of contaminants.

Compost mixing

Mix the manure and added material (e.g. by front-end loader) in a ratio that ensures that the blend is homogeneous and has an open structure to facilitate complete aeration of the composting pile while conferring a stable but loose structure to the finished product. The mixture should generally have a moisture content of about 50% to 60%, which can be maintained by rainfall, sprinklers or temporary covers.

Dairy manure does not produce unacceptable levels of odour during short periods of storage. Mixing and pile formation may be done once a week, but other wastes need daily incorporation. Spoilt feed and silage leachate have different characteristics. If week-long storage in uncovered stockpiles proves unsatisfactory, the stockpiles may be covered with a layer of bulking agent or mixed with the bulking agent and placed onto the composting pad daily.

pH

Compost should have a pH within the range of 5.0 to 8.0 to be compatible with plant growth and to avoid odour. A pH within the range of 5.5 to 6.5 is desirable if the compost is to be used as the sole component in a general potting medium, because within this range nutrients are most available to plants. Both saline and acidic conditions are not conducive to good composting, and ameliorants such as sulphur and lime are occasionally added to correct pH.

Moisture

Moisture loss is a good indicator of process activity, because evaporative cooling removes excess heat generated during the aerated stage of the composting process. In addition, moisture reduction is a common objective, because raw manure contains excess moisture, and a drier product is easier to handle, store and apply.

Temperature

The metabolic heat generated by the microorganisms elevates the temperature of the compost, so the control of temperature is an important aspect of composting. The temperature of the compost is a good indicator of the composting process activity; temperatures within 40 to 60 °C promote maximum biological activity. If moisture drops below 50%, the temperature will fall even if the composting process is incomplete. An elevated temperature (<60 °C) promotes rapid decomposition rates, and temperatures in excess of 55 °C for 3 days are effective in killing weed seeds. However, excessive temperatures can occur, and temperatures can exceed 60 °C, limiting microbial activity, delaying stabilisation and presenting a risk of spontaneous combustion. In the absence of heat removal via a decline in ambient temperature or good ventilation, and if oxygen is not limiting, composting temperatures can exceed 60 to 65 °C (and may reach 80 °C). Inadequate moisture can reduce efficiency in winter and generate excessive heat in summer, so water often needs to be applied, especially to avoid spontaneous internal combustion of a stockpile.

Nitrogen

Nitrogen is required by microorganisms for the breakdown of carbonaceous substrates. Insufficient N impairs the composting process, whereas excess N results in loss of N to the atmosphere by volatilisation of ammonia, which may cause noxious odours. An ‘ideal’ C:N ratio of 25–30:1 for the raw waste is acceptable. However, the nutrient status of compost is determined by the availability (to microorganisms) of the N and C, and therefore the required N content may different between manures generated by grazing animals and those from animals fed supplements.

The compost produced from dairy manure usually has a total N level of 1.0% to 1.5% by dry weight. This can be increased to around 3.0% by the addition of fertilisers and high-N wastes such as raw effluent. The increased N level in the waste will aid breakdown of plant fibres, so if they are available, add these wastes when composting manure. Under aerobic conditions, ammonia may be oxidised to nitrate and nitrite by nitrifying microorganisms. Oxidised N normally increases in concentration as a result of composting and is therefore sometimes used as an indicator of the success of the process.

Odour and appearance

A good indicator of compost maturity is the odour and appearance. A mature compost should be dark in colour and have a friable structure with an earthy odour. The presence of mycelium (fungal growth) is evidence of a poor composting process.

Other management techniques

Alternative techniques for managing the process are described in Keener *et al*. (2002); these include the use of temporary covers and aeration for quality control. The blending agents evaluated in this study were straw and sawdust. Although covers and aeration improved the quality of both straw and sawdust composts, researchers found little difference in the performance of the two blending agents, as further confirmed by Frederick *et al*. (2004).

The composting of dairy manure, with a high oxygen demand and a high potential for heat generation, requires management of both temperature and oxygen supply to promote rapid stabilisation, the selection of appropriate blending material, and a site that does not introduce undesirable organisms to the mix. Non-biodegradable materials like clay, silt and sand can impede the process by increasing the density and reducing the void ratio. The process can be further limited by the presence of disinfectants, antibiotics, herbicides and insecticides and by elevated levels of copper, zinc and chlorine.

Composting can be used as a way of disposing of animal carcasses. Murphy et al. (2004) successfully evaluated techniques for reducing carcasses into humus. This is now an accepted feedlot practice in Australia.

Compost use

Before use or sale, the compost can be passed through a screen to remove foreign objects. Depending on the structure and properties of the final product, it can be marketed as a soil conditioner, an ingredient for a potting mix or a complete potting mix or plant growth medium. The finished product can be sold in bulk to wholesalers or direct to the public as long as the source is specified. Although materials sold as soil conditioners or fertilisers must be registered and tested for quality and consistency, composts and mulches do not have this requirement. However, all marketed products need to comply with the Australian Standard for composts, soil conditioners and mulches (Standards Australia 2003).

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3.1 Nutrient budgeting

Dairy production systems typically require regular nutrient applications, especially of the macronutrients nitrogen (N), phosphorus (P), potassium (K) and sulphur (S), to meet nutrient removal rates of pastures and crops (DPI 2004, Gourley *et al.* 2007a). When nutrients are used in excess, they have the potential to significantly degrade air and water quality. The risk of nutrient pollution from a dairy farm increases when nutrient inputs exceed the amount leaving the farm in products (Gourley *et al.* 2007b). Total P and N inputs onto dairy farms, mainly in the forms of feed, fertiliser and N fixation by legumes, are usually much greater than the outputs in milk, animals and crops, so the surpluses tend to increase as farms intensify and stocking rates increase. In addition to off-farm environmental impacts, nutrient accumulation on dairy farms can result in unnecessary expenditure on feed supplements and fertiliser, and may reduce animal health and production (Gourley *et al.* 2007b).

A significant proportion of nutrients on a dairy farm can end up in the effluent (Gourley *et al.* 2007b). These nutrients provide a valuable resource and should, where possible, be used to replace nutrients removed from pastures and crops and to replace fertiliser (Gourley *et al.* 2007a, McDonald *et al.* 2005). The quantification of nutrients in effluent and their subsequent fate are important considerations in dairy effluent management. Farm nutrient budgeting tools are important tools to assess the risks associated with adverse environmental or production impacts that could result from nutrient deficiency or excess.

A nutrient budget, defined as an accounting approach to nutrient inputs, stores and outputs, can help manage nutrients by identifying production goals and opportunities for improvements in nutrient use efficiency, and thus reduce the risk of off-farm nutrient impacts (Gourley *et al.* 2007b). Nutrient budgeting for a dairy effluent management system is more specific than a whole-farm or farm-gate nutrient budget, as only the components of the effluent management system are assessed, such as manure collection and storage, nutrient redistribution, and crop or pasture nutrient uptake. This simple nutrient budget is a common and easy-to-calculate method that use readily available data at the farm scale and from sources that are likely to be fairly accurate. The nutrient budgeting allows for nutrients to be distributed in appropriate quantities for particular crops or pastures over sufficient areas of the farm. This provides a basis for minimising off-farm environmental impacts and efficient nutrient management (which reduces expenditure on feed supplements and fertiliser), and adverse impacts on animal health and production.

A farm nutrient budget does not normally try to directly quantify environmental losses such as P and N runoff, P and N leaching, denitrification or N volatilisation, as these are difficult to measure and are highly variable in space and time. The budgeting assesses nutrient accumulation and loadings and therefore environmental risks associated with the internal transformations, storages and distribution of nutrients across a farm. Nutrient budgets have also been found to be useful tools in improving farmer knowledge about nutrient flows and potential losses from their farms, and can influence fertiliser and manure management decisions. A detailed account of the advantages and limitations of nutrient budgeting is provided in Gourley *et al.* (2007b).

Nutrient importation

Nutrients are imported onto dairy farms principally through fertilisers and stock feeds, but can also be imported in animals, by nitrogen fixation, in bedding, in manure and in irrigation and rain water (Gourley *et al.* 2007b). The rate of nutrient importation will vary depending on the type of system; for example, a dairy farm which has an appropriate stocking rate, cuts its own hay and reuses all nutrients on farm would require lower imports of nutrients (in both feed and fertiliser), whereas a dairy farm with very high

stocking rates would be more likely to require higher rates of feed import and fertiliser application and thus will have a higher import of nutrients (McDonald *et al.* 2005).

The amount of nutrients imported through fertilisers can readily be quantified from the proportion of nutrients in the fertiliser and the fertiliser application rate. Although the amount of nutrients imported in stock feeds is often more difficult to quantify, Table 1 provides an indication of macronutrient concentrations typically imported onto a dairy farm in various feed types.

Table 1. Typical nutrient concentrations of stock feeds on a dairy farm (Helyar and Price 1999)**.**

Feed type	Nitrogen ($kg{\cdot}t^{-1}$ DM)	Phosphorus ($kg{\cdot}t^{-1}$ DM)	Potassium ($kg{\cdot}t^{-1}$ DM)	Sulphur ($kg{\cdot}t^{-1}$ DM)
Fodder				
Hay (cereal)	20	2	12	1.5
Hay (legume)	30	3	22	2
Hay (mixed pasture)	25	2.5	17	2.5
Grains				
Wheat and barley	17.5–21	2.8	4	2.2
Oats	19.8	3	4	2
Lupins	48–57	3.0–4.0	8.5–9.5	3.5
Other pulses	35–45	3.0–4.0	8.0–11.0	1.8–2.5

Nutrient concentrations in grains and forages can vary substantially and may have a large impact on the resultant nutrient budget outcomes (Gourley *et al.* 2007b).

Dairy effluent is often shandied with other water sources (such as irrigation or bore water). In this case, the quality of these other water sources, especially the impacts of salts and sodium, needs to be considered along with the effluent.

Nutrient distribution

Although uniform grazing management aims at an even distribution of manure across a farm, this is rarely achieved. Nutrients are concentrated in some areas when dairy cows are grazed in specific areas during day and night, when feedpads are used for part of the day, or when grazing regimes change with feed importation. Effluent and manure management systems should, as much as possible, manage the build-up of nutrients associated with any location where manure is concentrated, such as within laneways, the dairy shed and yards, on feedpads or in sacrifice paddocks. See chapter 1.2 ‘Characteristics of effluent and manure’ for details on the proportion of time that cows spend at the dairy.

Where significant amounts of nutrients are collected in dairy effluent, it is beneficial and more environmentally benevolent to spread these nutrients over areas that require nutrient increase (as based on soils analysis) rather than just over convenient paddocks adjacent to the effluent storage site. Effluent spread on convenient paddocks that are already high in nutrients will not increase pasture production and could therefore be regarded as a cost rather than a benefit. Where significant amounts of nutrients are collected in dairy effluent, it is beneficial to identify low-nutrient-status paddocks and apply the effluent there. The nutrients in the effluent can then be used to offset fertiliser costs. A sound nutrient budgeting system takes into consideration not only the nutrients being imported and exported on a whole-farm basis, but also the nutrients being transported within a farm (McDonald *et al.* 2005).

Effluent nutrient concentrations

The first step in nutrient budgeting is the quantification of the volume of effluent generated and collected and the concentration of nutrients in the effluent. The most

accurate method of quantifying effluent nutrient concentrations is site-specific data, however, as such data are rarely available, nutrient concentrations should be based on the information in chapter 1.2 ‘Characteristics of effluent and manure’. The nutrient concentrations shown in Table 2 (after Nennich *et al.* (2005) and based on a milk yield of 16.5 L·day^{-1}) have been used for the nutrient budgeting in this chapter.

Table 2. Nutrient concentrations in dairy cow manure used for nutrient budgeting (Nennich et al. 2005).

Nutrient	Nitrogen (g per cow day^{-1})	Phosphorus (g per cow day^{-1})	Potassium (g per cow day^{-1})
	393	63	178

Sulphur

Although S is an important macronutrient, data on S concentrations in dairy cow manure and on S removal rates in crops are limited. As a result, nutrient budgeting for S without site-specific data is inaccurate and can be misleading. This area requires more research (Reuter and Robinson 1997).

Trace elements

The impact of trace elements from dairy effluent is marginal. McBride and Spiers (2001) found that trace element levels in both liquid and solid dairy manures were typically low, except where feed additives were used. Feed additives generally increased levels of copper and zinc. Trace elements are discussed further in chapter 3.5 ‘Trace elements’.

Nutrient quantification

In quantifying the nutrients collected in a dairy effluent system, you need to apportion the data in Table 2 according to the time the cattle spend on areas from where the effluent is collected. Ideally the time spent on those areas should be estimated to support future management. Alternatively, a less accurate estimate can be obtained from the rough rule of thumb that dairy cattle spend 10% to 15% of their day on an area from which the effluent is collected (see chapter 1.2 ‘Characteristics of effluent and manure’). An example is provided in Table 4.

The concentrations of nutrients within the effluent system can change. Although the magnitude of change is variable and depends on a range of factors, only N is generally lost throughout collection and conveyance. Other than N nitrogen, if storage ponds are correctly lined, there should be no loss of nutrients from the pond even after stirring. Nutrient locations and losses within an effluent pond system are shown in Table 3.

N within effluent is highly mobile and is lost to the atmosphere through volatilisation and denitrification at all times (Kruger *et al.* 1995). About half of the N typically lost through volatilisation. More information on N conversions and losses from the effluent system is discussed in chapter 3.2 ‘Nitrogen’.

Solids, sludge and liquid effluent

Nutrient budgeting assumes that solids from separation and sludge from effluent ponds are applied to land along with effluent. Although effluent from all of these sources may not be applied simultaneously, it is assumed that these sources are all applied to the same sites. This would typically occur over a number of years.

Paddock-specific nutrient budgeting can be based on the application of solids from a solids separator or of liquid effluent from a storage pond. Site-specific data yielded from analysis of effluent samples provide the most accurate way of quantifying nutrient levels for this type of budgeting. However, in the absence of such data, the proportions detailed in Table 3 can be used. Pond stirring can redistribute nutrients throughout the

storage, but these nutrients soon settle over time, and some become locked up in sludge (NSW Dairy Effluent Subcommittee 1999). More information on sludge management is discussed in chapter 2.8 ‘Desludging and pond closure’.

Table 3. Nutrient locations and losses within an effluent pond system (NSW Dairy Effluent Subcommittee 1999).

	N (%)	P (%)	K (%)
Effluent	30	40	90
Sludge	20	60	10
Loss	50	–	–

Table 3 shows that the 50% of the N excreted will be available for reuse on land. The calculations in Table 4 allow for this.

Table 4. Quantity of nutrients generated and collected per year for a 300-cow dairy herd, milked twice a day and supplementary-fed on a feedpad.

Nutrient	Nitrogen	Phosphorus	Potassium
Nutrient produced in manure from Table 2 (g per cow day^{-1})	393	63	178
Hours spent per day on areas from which effluent is collected			
Laneways	0.5		
Dairy shed and yards	3.0		
Feedpad	3.0		
Total hours	6.5		
Proportion of day effluent collected	0.271		
Nutrient collected (g per cow day^{-1})	107	17	48
Nutrient available for reuse from Table 3 (g per cow day^{-1})	53	17	48
Total nutrient (kg) per day [a]	15.9	5.1	14.4
Total nutrient (t) per year [b]	4.85	1.56	4.39

a: Based on 300 cows.
b: Based on 305-day lactation.

Nutrient use and export

Nutrients are utilised on dairy farms in the production of pastures and crops. The level nutrient removal will vary with crop type, yield and growing conditions. Table 5 indicates the amounts of N, P and K removed by a range of crops.

Table 5. Nutrient removal (where product is removed from the site) in particular crops.

Crop (yield, wet t·ha^{-1})*	N removal (kg·ha^{-1}·y^{-1})	P removal (kg·ha^{-1}·y^{-1})	K removal (kg·ha^{-1}·y^{-1})
Barley (3.5 t)	168	27	140
Lucerne hay (7.5 t)	209	19	141
Maize silage (50 t)	165	65	206
Millet (9 t)	280	45	186
Oats (3.5 t)	168	27	140
Perennial pasture for hay (15 t)	150	18	80
Perennial ryegrass for hay (15 t)	200–250	25–40	200
Wheat (2.8 t)	208	27	150
Sorghum grain (9 t)	280	45	186
Triticale (2.8 t)	168	27	140
Dairy pasture (10 t DM ha^{-1})	400	40	200

*Adjust according to anticipated or measured yields.

3.1 Nutrient budgeting

Nutrients can be exported from a dairy farm through products (milk, meat, animals, manure, crops) and through losses such as runoff, leaching and N volatilisation (Gourley *et al.* 2007b). However, effluent nutrient budgeting typically does not consider the whole-farm nutrient cycle and thus does not assess nutrients exported off farm. Rather, effluent nutrient budgeting is usually limited to determining the appropriate quantities of nutrients to be distributed for particular crops or pastures, and indicates an area of land sufficient to assimilate the collected nutrients.

For those interested in undertaking whole-farm nutrient budgeting that takes into account nutrients exported as milk, Table 6 indicates average quantities of macronutrients found in milk.

Table 6. Macronutrients in milk (Helyar and Price 1999).

	N	P	K	S
kg of nutrient per 10 000 L milk	42	10	14	3.2

Nutrient budgeting calculations

Once the total annual quantity of nutrients collected by the effluent management system has been calculated (as in Table 4), the next step is to calculate how these nutrients can be utilised in appropriate quantities on crops or pastures. The effluent nutrient budget sets a minimum area of land required to utilise the collected nutrients while minimising the risk of excess nutrient loss through runoff, leaching or volatilisation. The process of land application should be governed by a limiting constituent analysis (Midwest Plan Service 1985), which determines the limiting nutrient to ensure that it is not over-applied to land; that is, the limiting nutrient loading stays at or below the maximum requirements for a particular crop or pasture. This is done by determining the typical expected yield of the intended crop or pasture and calculating the nutrient removal by this crop or pasture at that yield from data such as that provided in Table 5. This information is then compared with the total annual quantity of nutrients collected by the effluent management system (as calculated in Table 4) to determine the area of land required to utilise the nutrients collected. This process is detailed in the worked examples below. A further calculation will then determine the required effluent loadings as nutrient load: $kg{\cdot}ha^{-1}$ for solids or $ML{\cdot}ha^{-1}$ for liquid.

Nutrient budgeting inputs and considerations

The simple effluent nutrient budgeting described above needs as inputs:

- the nutrient levels within the effluent (Table 2)
- the fate of the collected nutrients to determine availability (Table 3)
- the total annual quantity of nutrients generated (Table 4)
- the proposed or existing crop on the land application area
- estimated crop yield (district average or from past experience)
- crop nutrient removal rate (such as data in Table 5).

We also need to consider:

- crop water requirement (typically determined from climatic water budgeting as detailed in chapter 3.9 ‘Hydraulic application rate and scheduling’)
- the proposed dilution or shandying rate (if any)
- achievable effluent loadings (e.g. $kg{\cdot}ha^{-1}$ for solids or $ML{\cdot}ha^{-1}$ for liquid)
- the areas of land over which effluent can realistically be applied (may vary between solids and liquids)

- climatic predictions, landform and soil type and the associated risk of runoff or leaching
- existing soil nutrient concentrations
- type and rate of recent (past 12 months) or proposed fertiliser applications.

Any imbalance in nutrients required for optimum production can be made up through fertiliser applications. Similar techniques can be used with maximum loadings for particular soils. If so, refer to state EPA guidelines for maximum soil nutrient loadings. Where P is the limiting constituent, calculate the P retention capacity of the soil to indicate the lifespan of the reuse area before soil P saturation occurs. This is discussed in chapter 3.3 ‘Phosphorus’.

Keep in mind salt and sodium loadings when applying effluent to land, including any salts present in irrigation or bore water also applied to the land. See chapters 3.6 ‘Salinity’ and 3.7 ‘Sodicity’.

Worked examples

Example 1—Total nutrient budgeting

This nutrient budget is based on the total quantity of nutrients collected per year and on spreading those nutrients over an appropriate area of land as governed by the limiting constituent analysis (Midwest Plan Service 1985) detailed above. This budget requires:

- the quantity of nutrients generated (calculated as in Table 4)
- the proposed or existing crop
- the estimated crop yield (district average or from past experience)
- the crop nutrient removal rate (such as in Table 6).

Table 7 details a nutrient budget for an example dairy farm with effluent nutrient loadings taken from Table 4 and the nutrient removal rate by perennial pasture taken from Table 6 and adjusted for slightly lower yields.

Table 7. Example 1—Total nutrient budget, 300 cows with 27% of manure collected.

	Nitrogen	Phosphorus	Potassium
Total nutrient collected from Table 4 ($t{\cdot}y^{-1}$)	4.85	1.56	4.39
Proposed crop	Ryegrass clover perennial pasture		
Crop nutrient removal from Table 5 ($kg{\cdot}ha^{-1}{\cdot}y^{-1}$)	400	40	200
Crop yield (t DM ha^{-1})	10		
Proposed yield (t DM ha^{-1})	8		
Nutrients removed in proposed crop ($kg{\cdot}ha^{-1}{\cdot}y^{-1}$)	320	32	160
Area required to utilise all nutrients (ha)	15.2	48.8	27.4

From Table 7, we can conclude that the effluent generated in 1 year should be applied to 49 ha of land to utilise all of the P collected. For optimum production, top-up applications of K and N will be needed.

Example 2—Liquid effluent nutrient budgeting

This nutrient budgeting example is based on nutrient levels within storage ponds derived from site-specific analysis. Rather than directly determining the area of land required (as in Example 1 above), this budget is based on determining the appropriate volume of effluent to apply per ha of land and includes scenarios for shandying effluent with irrigation water. The volume of effluent to be reused then determines the areas of land required as governed by the limiting constituent analysis (Midwest Plan Service

1985). As this budgeting considers the liquid effluent only, solids and pond sludge would need to be applied on separate land.

This budget requires:

- the nutrient levels within the liquid effluent (from analysis)
- the proposed or existing crop
- the estimated crop yield
- the crop water requirement
- the proposed dilution rate(s)
- the crop nutrient removal rate (from Table 5)
- the loadings of nutrients per hectare (kg·ha^{-1} for solids or ML·ha^{-1} liquid).

It is also advantageous to know the existing soil nutrient concentrations. Table 8 details an example nutrient budget for an irrigation farm with measured effluent nutrient concentrations and growing an irrigated maize crop, with the crop nutrient removal rate from Table 5.

Table 8. Example 2—Liquid effluent nutrient budget.

	Nitrogen	Phosphorus	Potassium
Liquid nutrient concentration (measured) (mg·L^{-1})	25.6	14.3	17.8
Proposed crop	Irrigated maize silage		
Crop nutrient removal from Table 5 (kg·ha^{-1}·y^{-1})	165	65	206
Crop yield (t·ha^{-1})	50		
Proposed yield (t·ha^{-1})	60		
Nutrients removed in proposed crop (kg·ha^{-1}·y^{-1})	198	78	247
Crop water requirement (ML·ha^{-1})	10.0		
Proposed dilution rate (ratio)	1 part effluent to 1 part water		
Proposed annual application (ML·ha^{-1})	5.0 ML effluent : 5 ML irrigation water		
Nutrient concentration in irrigation water (mg·L^{-1})	12.8	7.15	8.9
Nutrient applied to crop (kg·ha^{-1})	128	71.5	89
Nutrient excess (+) or deficit (–) (kg·ha^{-1})*	–70	–6.5	–158

*Where an excess occurs, further dilution of the effluent is required. Alternatively, a different crop could be selected that needs less water or removes more nutrients.

From Table 8, we can conclude that the liquid effluent could be applied to the crop at a rate of 5 ML·ha^{-1} (over the growing season) and the maize would still utilise the P applied. In this case P is the limiting constituent, and higher applications of effluent (such as through a lower dilution ratio) would apply P in excess of crop requirements. In this scenario above, for optimum production, top-up applications of K and N would be needed.

Nutrient budgeting computer programs

Various spreadsheets are available to assist with effluent nutrient budgeting. Some are detailed and are more suitable for research, such as DAIRYBAL, from the Queensland DPI (McGahan *et al.* 2004), which determines the manure output of a dairy herd from the rations fed to the cattle and the pastures or forage crops grazed. The model assesses the nutrient mass balance on the effluent application areas to determine whether the proposed cropping or pasture management practices are environmentally sustainable in terms of nutrient loading. However, for a typical farm-based nutrient budget, simpler spreadsheets provide the area of land required over which to apply (and therefore reuse) the nutrients generated and collected in the effluent management system. Such a model has recently been developed by the Victorian DPI (Scott McDonald, pers. comm., DPI Vic., 2007).

When using these models, take care that the inputs are accurate. In addition, interpretation of the outputs, which can be complex, often requires careful consideration to ensure that the outputs are used appropriately and kept in context.

Nutrients and the environment

Nutrient surpluses at the paddock level and the subsequent accumulation and losses to the broader environment are often complex and highly variable in both space and time (Gourley *et al.* 2007b). Details on specific nutrients and environmental issues relating to these are provided in chapters 3.2 'Nitrogen', 3.3 'Phosphorus', 3.4 'Potassium' and 3.5 'Trace elements'.

In general, the impact of effluent reuse practices on farms and their surrounding environment through surface runoff, leaching or volatilisation causes concern. Dalal *et al.* (2003) describe the effect of greenhouse gases derived from losses of N from farms. Eutrophication—the enrichment of nutrients—is caused by a build-up of nutrients and can lead to blue-green algal outbreaks (Drewry *et al.* 2006). Dairy farms can contribute to problems on account of the high levels of nutrients that can run off and leach. Runoff is common after rainfall or irrigation, and carries with it nutrients. Fleming and Cox (2001) state that 98% of total nutrient loss over a 3-year period came from overland flow. Average annual losses from dairies can be as high as 22.8 kg total N ha^{-1}, 10.0 kg total P ha^{-1} and 43.1 kg K ha^{-1} (Holz 2007). Although farm nutrient losses often depend on climate, hydrology, soil and landscape (which are often out of the manger's control), nutrient runoff can be minimised through careful timing and application of effluent and through good soil conservation practices such as contour banks, minimum or zero tillage and strip cropping. As long as these practices are appropriate for the climate or location, they can help to minimise soil and nutrient losses. Although runoff from effluent reuse areas should flow into farm drainage structures and end up in recycling ponds and thus not leave the farm, the construction of runoff collection dams and ponds is currently not permitted in some catchments under water management plans.

Further information on the environmental impacts of dairy nutrient management and nutrient losses can be assessed by using the Farm Nutrient Loss Index (FNLI), developed in the Better Fertiliser Decisions project (Gourley *et al.* 2007a). This project collected comprehensive information to improve fertiliser decisions for grazing industries across Australia. It compiled and interpreted results from pasture fertiliser experiments and information on nutrient loss processes in all pastoral regions in Australia. It revealed the relationship between soil test results and pasture response and gave critical soil test values for P, N and S at regional, state and national scales, and by soil characteristics such as soil texture and P buffering index. The FNLI is a computer-based decision support tool used to assess the risk of nutrient loss from the paddock to the off-farm environment. It predicts the relative risk of P and N loss processes and is designed to assist farm advisors, in conjunction with farmers, to make informed nutrient management decisions. The FNLI takes into account the pathways of nutrient loss relevant to Australian pasture-based industries:

- runoff across the soil surface
- drainage past the root zone
- lateral flow through subsurface layers in the soil profile
- emission of greenhouse gases.

For more details on the FNLI and the Better Fertiliser Decisions project, refer to Gourley *et al.* (2007a).

Further research

More research and model development is required to achieve a more robust and thorough nutrient budgeting tool for Australian dairy farms (Gourley *et al.* 2007b) or an overall farm nutrient management model that considers environmental impacts and farm production, such as the Overseer Nutrient Budgets used in New Zealand (Mathew Redding, pers. comm. QDPI, 2007). Gourley *et al.* (2007b) suggest that a nutrient budgeting tool must cover not only basic input and output of nutrients on dairy farms, but also:

- diets fed to dairy cows
- manure nutrient loads
- manure forms
- manure nutrient collection, storage and redistribution
- productive and non-productive areas
- soil testing data.

Gourley et al. (2007b) argue that this will identify excess accumulation of nutrients within particular management units, quantify relative nutrient efficiencies, and hence identify the opportunities to improve nutrient management decisions by dairy farmers and advisors, and enhance environmental outcomes. Such a nutrient budgeting tool needs to:

- identify and quantify key nutrient inputs, outputs and stores (e.g. feed, manure) and nutrient surplus and efficiencies at both the farm gate and more the farm system levels
- define the uncertainties in nutrient budget calculations and predictions
- identify and quantify nutrient distribution within the dairy farm and nutrient losses off the farm
- integrate nutrient budgets at the field level with recommended Australian soil test targets and use this information to support fertiliser recommendations
- provide an effective assessment of costs and benefits resulting from current nutrient management practices
- establish appropriate targets for permissible surpluses and potential nutrient efficiencies at the whole-farm and component levels
- recommend management practices which will improve nutrient budgets and nutrient use efficiencies.

The results of the dairy farm nutrient budgeting provided above should be used to assess the environmental sustainability of a dairy farm. The area of land required to utilise the limiting nutrient must be available for the intended purpose. On low-intensity dairy farms where no feed is imported, this is typically readily achieved. On high-intensity, high-stocking-rate dairy farms that import significant volumes of feed and often use a feedpad system, the area of land required to reuse the nutrients collected in the effluent management system can be substantial and needs to be available in order for the enterprise to continue or proceed. The option of exporting effluent in liquid or solid forms, such as to surrounding farms, could be considered.

Despite the complexities associated with nutrient management on a dairy farm, effluent nutrient budgeting results in improved nutrient use efficiency and reductions in nutrient surpluses at the farm level; improved environmental performance at the catchment or broader scale; and greater confidence among farmers, advisors and policy makers in the use of nutrient budgeting to enhance nutrient management and environmental performance on dairy farms.

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3.2 Nitrogen

For general management guidelines pertinent to all nutrients, see chapter 3.1 'Nutrient budgeting'. Most issues pertinent to N are dealt with in that chapter, but some additional issues specific to N are dealt with here.

N is a nutrient found in high quantities in dairy effluent. N is an essential nutrient for dairy pasture and crop production, and dairy effluent can be used to replace some of the N required within these systems. N in dairy effluent is subject to many conversions and losses throughout handling, storage and land application, and these are difficult to quantify. Because N is a highly mobile nutrient, it must be managed carefully to avoid adverse environmental impacts.

More intense scrutiny of N management within dairy farms is occurring owing to off-site impacts. Volatilisation and denitrification cause some N to be lost as either nitrous oxide (N_2O) or ammonia gas (NH_3). Nitrous oxide, a greenhouse gas, is implicated in climate change (AGO 2007, Dalal *et al.* 2003, Thomas *et al.* 1999). N from surface runoff has been found to contribute to eutrophication—the nutrient enrichment of ecosystems—which can lead to outbreaks of blue-green algae (Harrison 1994, Robson and Hamilton 2003). Leaching of nitrate (NO_3^-) has also polluted groundwater owing to high concentrations of applied N, typically on free-draining soils (Cameron and Di 2004, Silva *et al.* 1999).

Forms of nitrogen and nitrogen cycling

The cycling of N is complex and highly variable in effluent management systems (Gourley *et al.* 2007b). N within dairy effluent is found in both organic and inorganic forms, the latter as ammonium (NH_4^+) and nitrate. Dairy effluent typically contains 60% to 85% organic N, and initially only a small proportion of N occurs in inorganic form (Barkle *et al.* 2000). When effluent is applied to land, N can undergo a number of changes, including:

- immobilisation of inorganic forms by plants and microorganisms to form organic N compounds
- mineralisation—the decomposition of organic N to ammonium
- nitrification—the oxidation of ammonium to nitrite (NO_2^-) and then to nitrate
- denitrification of nitrate to nitrous oxide and N gas (N_2).

The processes involved when dairy effluent is applied to land (Kruger *et al.* 1995) are explained here. On the application of dairy effluent to land, the amount of N mineralised or immobilised depends on the form of organic matter present, temperature and soil moisture. Under suitable conditions, microbial populations increase rapidly. This provides a large sink of N for use in cell synthesis. Ammonium can also be used by the microbial biomass provided there is a carbon source in the effluent to support growth. This incorporation is termed immobilisation, the opposite of mineralisation; the balance between the two processes is determined largely by the C:N ratio of the added material. As a rule of thumb, if C:N > 25, there is net immobilisation, because sufficient carbon is present to stimulate microbial growth such that all the N added in the effluent is incorporated into the microbial cell structure. It is important to realise that the decomposition of organic material is driven by the demand of the soil microflora for C as an energy source and building block for new cell growth. The release of N (as NH_4^+) and P and other inorganic material from the organic matter is only incidental to this microbial growth process. Nitrification is an aerobic process in which the relatively immobile ammonium form is transformed into nitrate, which can be readily leached from soil. Temperature and oxygen supply govern the rate of nitrification. Under aerobic, warm conditions there is almost complete conversion of ammonium to nitrate in the

surface soil within a few days of effluent application. Denitrification occurs under anaerobic conditions: oxygen is in short supply, so bacteria use nitrate or nitrite as a source of oxygen and produce N_2O and N_2 gases, which are lost from the soil.

Nitrogen losses

Around half of the N in fresh faeces and urine may be present as ammonia or be converted to ammonia shortly after excretion. This ammonia is very volatile, and unless it is absorbed by, or reacts chemically with, some substance, most of it escapes into the air. This process continues during treatment and storage. High temperatures and high pH increase ammonia loss. Loss of N from effluent increases with storage time. Long-term storage systems such as ponds have the greatest N losses. In solids-separation systems, about 10% of total N is retained in the solids. Significant amounts of N are commonly lost from most effluent treatment systems. This may be regarded as an advantage or a disadvantage, if the objective is effluent reuse with minimum N pollution hazard or efficient use of the fertiliser N content. More than half of total N excreted and most of the potential volatile N is present in urine. Any urine N that dries on concrete surfaces will escape to the air. If manure is stored as a liquid or absorbed in bedding, N losses are reduced (Kruger *et al.* 1995).

Nitrogen fixation

N fixation by legumes may be an important N input in both pasture and mixed-cropping dairy operations (Gourley et al. (2007b). In pasture systems that include legumes, N input from fixation can vary between 10 and 270 kg N $ha^{-1}{\cdot}y^{-1}$ but is typically between 80 and 100 kg N $ha^{-1}{\cdot}y^{-1}$. The amount of N fixed from the atmosphere by legumes is difficult to measure directly owing to spatial and temporal variability and the complexity of measurement techniques. Consequently, fixed values or ranges are often used, or N fixation is predicted using established algorithms, which are often incorporated into decision support tools and models.

Nitrogen in dairy effluent

N conversions and losses from dairy effluent vary depending on the amount excreted from animals, exposure to the atmosphere before suspension in water, the time the effluent is held in storage ponds and the method of land application. Accordingly, these conversions and losses vary significantly between dairy farms and are difficult to quantify, so data must therefore be seen as indicative only.

Excreted nitrogen

The amount and characteristics of animal manures excreted are dealt with in detail in chapter 1.2 ‘Characteristics of effluent and manure’. A dairy cow excretes approximately 3.2 kg of N per lactation within the area of the shed and yards (DPI 2005). However, this should be used as a guide only, as it varies considerably with location and feed type. The inorganic portion of this N is subject immediately to volatilisation and denitrification.

Effluent storage for nitrogen retention

Provided that storage ponds are lined correctly and seepage losses are negligible, N within storage ponds is generally converted into gaseous forms. Mason (1996) found that in a dairy effluent pond, N was lost mainly by volatilisation, at an average removal rate of 0.75 g ammonia $m^{-2}{\cdot}day^{-1}$. In addition to volatilisation, organic N in dairy effluent slowly mineralises into inorganic forms within storage ponds at a rate dependent on temperature, regardless of oxygen status (Zhao and Chen 2003). The amount of N in dairy effluent ponds varies with depth (McDonald *et al.* 2005) and between farms (see

chapter 2.3 ‘Anaerobic, aerobic and facultative ponds’ for typical ranges in concentration). As a result, Waters (1999) recommends analysis of the final effluent stream to be applied to land in each individual case.

Land application

The amount of N applied to land in dairy effluent depends on solids separation, the degree of aeration or agitation of storage ponds, the storage period and effluent pH. Applying dairy effluent to land can stimulate mineralisation and nitrification, resulting in a significant increase in soil nitrate concentrations (Zaman *et al.* 1998), but also significant N volatilisation (Chastain and Montes 2004). In the latter case, ammonia is released to the atmosphere; N losses of 25% have been recorded (Carey *et al.* 1997). Application methods and evaporation rates have not been found to significantly influence these processes (Chastain and Montes 2004).

Nitrogen uptake by pasture

Typically, the mineralisation of organic N in conventional dairy systems is less than the requirements of a standard dairy pasture, even where atmospheric N fixation by legumes contributes, so N applications are normally required (NSW Dairy Effluent Subcommittee 1999, Price 2006). Between 20% and 40% of effluent-applied N was utilised by pasture in the short term (Carey *et al.* 1997, Di *et al.* 2002), but the remaining N can take up to 3 years to become available to plants (NSW Dairy Effluent Subcommittee 1999).

Surface runoff of nitrogen

The N in dairy effluent applied to land is vulnerable to runoff losses facilitated by rainfall or irrigation, typically in the form of ammonium (Smith *et al.* 2001). Although N losses can be minimised by effluent incorporation through ploughing, runoff losses are typically controlled sufficiently through sound soil conservation practices such as vegetation retention, surface water runoff interception and sediment trapping (NSW Dairy Effluent Subcommittee 1999). The risk of N loss from dairy effluent application areas can be assessed through use of the Farm Nutrient Loss Index (FNLI) (Gourley *et al.* 2007a).

Nitrate leaching

Nitrate is susceptible to leaching and is potentially hazardous to groundwater supplies used for human consumption. The rate of leaching is highest in free-draining soils. Cameron and Di (2004) found that in addition to leaching of the susceptible nitrate N, ammonium and organic forms can also be lost through this mechanism. The land application of N therefore requires careful management in free-draining soils. Research has shown that the urine component of dairy effluent supplies most of the leached nitrate (Silva *et al.* 1999), and that splitting effluent applications (into multiple, smaller applications) can reduce N leaching losses (Cameron and Di 2004). The leaching of urine during grazing is of greatest concern, as a significant proportion of N ingested is excreted, typically back onto pasture, where N application rates far exceed the ability of the pasture to utilise the N (Cameron and Di 2004, Haynes and Williams 1993, Whitehead and Raistrick 1993).

Nitrogen loadings

Any N deficit in a dairy pasture or cropping system is typically rectified through fertiliser applications timed to coincide with peak N demand. Careful effluent and N fertiliser application is required, as the inorganic N forms are relatively mobile within soils and are readily taken up by a crop or pasture, volatilised or leached, and the organic portion of N is released slowly over time. In addition, N surplus can occur on dairy pastures

where effluent has been applied over many years, as total organic N levels can accumulate.

Nitrogen management

Most Australian soils are naturally low in N, and most agricultural pastures and crops, with the exception of legumes, require N applications to attain optimum production. N is readily taken up by plants and is required in significant amounts at certain critical growth stages. Owing to the transient nature of N, applications are typically distributed throughout the growing season and are often timed to meet peak demands. A quick response and regular applications are typically required to rectify N deficits. Soil analysis for N is not a reliable predictive tool for N management as there is no reliable soil test for N (Gourley *et al.* 2007a). Other indicators such as leaf analysis, plant symptoms and projected crop requirements (e.g. based on N removed in produce) are more reliable (Strong and Mason 1999). It is difficult to indicate typical annual maintenance rates for a dairy pasture, as the requirements vary considerably depending on management and location. However, annual application rates up to 200 $kg{\cdot}ha^{-1}{\cdot}y^{-1}$ are typical, and rates up to 300 $kg{\cdot}ha^{-1}{\cdot}y^{-1}$ are acceptable on high-yielding kikuyu (*Pennisetum clandestinum*) pastures. These applications would typically be applied in split dressings of 20 to 60 $kg{\cdot}ha^{-1}$ at optimum times for yield maximisation.

Monitoring nitrogen

Details on monitoring N throughout a dairy effluent management system are provided in chapter 7 ‘Monitoring and sampling’.

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3.3 Phosphorus

For general management guidelines pertinent to all nutrients, see chapter 3.1 'Nutrient budgeting'. Most issues pertinent to P are dealt with in that chapter, but some additional issues specific to P are dealt with here.

As P is essential for rapid pasture and crop growth and is not naturally abundant in Australian soils (Moody and Bollard 1999, Price 2006), the significant amounts found within dairy effluent can be used to supply pasture and crop P requirements (McDonald *et al.* 2005, Wang *et al.* 2004), thus reducing fertiliser inputs and bringing substantial savings (Skerman *et al.* 2006).

Although P is relatively immobile in soils, it can be lost in surface runoff or by leaching, particularly in association with rainfall or irrigation (McCaskill *et al.* 2003). The P collected through a dairy effluent management system therefore needs to be managed prudently.

Phosphorus in dairy effluent

Phosphorus collection

Minimal P is lost throughout collection and conveyance of dairy effluent, and it can be assumed that all P collected will be available for reuse, providing that solids from separation, sludge from effluent ponds and liquid effluent are all considered. The quantity of P within dairy effluent will vary with location and feed type, and particularly with dietary P forms and levels (Ebeling *et al.* 2003). The amount of P entering an effluent storage pond depends on the amount of solids separated from the effluent stream and varies considerably. See chapters 2.3 'Anaerobic, aerobic and facultative ponds' and 2.8 'Desludging and pond closure' for typical P concentrations in dairy effluent and sludge. Although these chapters can provide a guide, it is more accurate to analyse the dairy effluent in each individual case (Waters 1999).

Phosphorus uptake by plants

P in dairy effluent is excreted in both organic and inorganic forms. Organic P (unavailable to plants) becomes inorganic (available) through mineralisation. This process varies with both time frame and output (Moody and Bollard 1999). The removal of P from soils is almost entirely due to plant uptake and harvest, which depends on nutrient availability and soil pH. Sites with a long-term history of dairy effluent application, in particular solids application, typically have adequate soil P levels. Intensive hay production can significantly reduce available P levels in soil. However, the interactions between available P and the total soil P pool is complex, and is discussed in more detail below.

Phosphorus losses

P is readily fixed to soils, especially clayey soils, and is far less mobile within soils than other nutrients, moving very little from initial placement (Price 2006). The likelihood of leaching is highest in coarse, sandy, well drained soils, but leaching can also occur where loadings are excessive or through bypass flow mechanisms (Redding 2001). Excess P may be retained by soil and only slowly released through diffuse surface runoff processes, or alternatively lost in significant amounts during episodic erosion events (Gourley *et al.* 2007b, Nash and Murdoch 1997). The quantity of P required to reduce water quality is very small, especially in comparison to N, so the movement of very small quantities of P off site can have adverse environmental impacts (Nash and

Murdoch 1997, Redding 2001). When effluent is applied to paddocks, surface runoff is the most likely method of P loss in both soluble and particulate forms, typically in association with rainfall or irrigation (Drewry *et al.* 2006, Holz 2007, McCaskill *et al.* 2003, Nash and Murdoch 1997, Redding 2001). Higher concentrations of P in surface runoff are associated with sites with a greater use of P fertiliser or effluent application, especially where soil incorporation does not occur (Fleming and Cox 2001, McCaskill *et al.* 2003). Fleming and Cox (2001) found that 98% of P loss over a 3-year period was due to overland flow. To help minimise the risks of P loss in surface runoff, careful monitoring of soil and effluent P levels is required, along with responsive management. In addition, surface runoff should flow into drainage lines and recycling ponds so that no contaminated runoff leaves the farm (Drewry *et al.* 2006, Nash and Murdoch 1997). The risk of P loss from effluent application areas can be assessed through use of the Farm Nutrient Loss Index (FNLI) (Gourley *et al.* 2007a)—see chapter 3.1 'Nutrient budgeting'.

Phosphorus management

Phosphorus in soils

Most Australian soils are naturally low in P, so pastures and crops require P applications for optimum production (Moody and Bollard 1999). Although much of the P pool remains unavailable as fixed or adsorbed P (Figure 1), P can become available in the soil solution from plant and microbial processes (Barrow and Shaw 1975, Moody and Bollard 1999). Most P is taken up in the upper few mm of the soil profile, and the availability for uptake varies with the soil P buffering index (PBI), soil texture, temperature, time, pH, rainfall, plant species, management, microbial populations and mineralisation rates (Barrow and Shaw 1975, Burkitt *et al.* 2002, Gourley *et al.* 2007a, Moody and Bollard 1999).

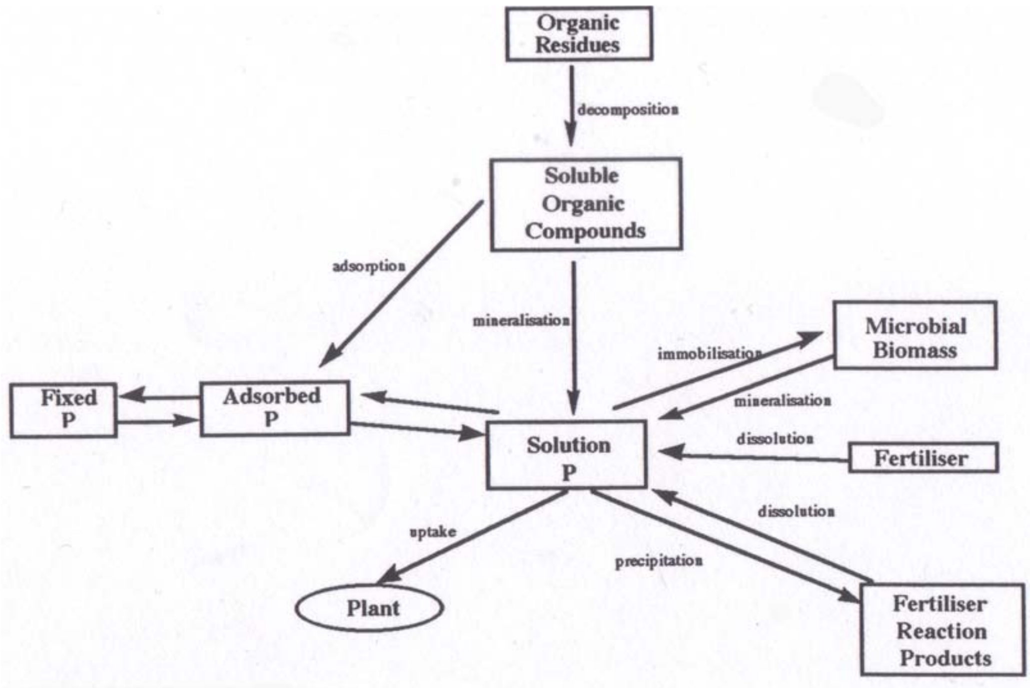

Figure 1. Phosphorus cycling in soils (Moody and Bollard 1999).

Phosphorus and soil analysis

P in soils is typically measured as plant-available P by a range of methods, the most common being Colwell P and Olsen P. Both tests are relatively accurate and reliable, but soil-available P can be affected by a range of factors, as listed above (Moody and

Bollard 1999). These soil analysis results, especially Colwell P, need to be interpreted in association with an estimate of the soil's P-fixing capacity (Gourley *et al.* 2007a). Although soil texture or other measures have long been used as surrogates for soil P-fixing capacity, the recently developed PBI is now the national standard (Burkitt *et al.* 2002, Gourley *et al.* 2007a).

Soil phosphorus levels

Soil analysis for P is a reliable method of assessing soil P requirements. The soil should be analysed before effluent or fertiliser is applied to assist in determination of appropriate P loadings (Gourley *et al.* 2007a, Moody and Bollard 1999). Gourley *et al.* (2007a) found that critical available P levels for pastures measured as Olsen P were applicable Australia-wide regardless of region, soil texture or PBI. The critical soil-available P level to achieve 95% of maximum pasture production across Australia, measured as Olsen P, was 15 $mg{\cdot}kg^{-1}$.

Gourley *et al.* (2007a) also found that critical available P levels for pastures measured as Colwell P depended significantly on PBI, but not on region or soil texture. They developed an equation enabling determination of critical Colwell P values when the PBI of a soil is known and used this to calculate critical Colwell P values for commonly used PBI categories, as detailed in Table 1.

Table 1. From Gourley *et al.* (2007a).

Predicted critical Colwell P soil test values for standard PBI categories, derived from the national data set.

PBI category		Critical value for mid point of PBI category (range)[1]
<15	Extremely low	23 (20 – 24)
15-35	Very very low	26 (24 – 27)
36-70	Very low	29 (27 – 31)
71-140	Low	34 (31 – 36)
141-280	Moderate	40 (36 – 44)
281-840	High	55 (44 – 64)
>840	Very high	n/a[2]

[1] Critical Colwell P value (mg/kg) at the mid-point of PBI category. Values in parenthesis are critical Colwell P values at the lowest and highest PBI values within the category.

[2] Insufficient data to derive a response relationship.

Phosphorus fixation and buffering

The amount of phosphorus sorbed in a soil and the subsequent P available for plant growth will largely depend on the soil type and its sorption capacity (Burkitt *et al.* 2002, Gourley *et al.* 2007a, Kruger *et al.* 1995, Moody 2007, Slattery *et al.* 2002). P sorption capacities can be obtained by using the isotherm method (DPI 2001, Kruger *et al.* 1995); however, as this method is detailed, time consuming and costly, the PBI is the preferred method of assessing a soil's propensity to retain P (Burkitt *et al.* 2002, Gourley *et al.* 2007a). This propensity should be assessed in conjunction with Colwell P or Olsen P soil analysis, and the P requirement should be interpreted according to the PBI range, as detailed in Table 1 (Burkitt *et al.* 2002, Gourley *et al.* 2007a, Moody and Bollard 1999).

3.3 Phosphorus

The PBI can be used to indicate the amount of P that could theoretically be sorbed by the portion of the soil profile that effluent will infiltrate before significant leaching of P occurs. Experience and research on soils treated with piggery effluent indicate that the actual amount of P sorbed in the field before leaching occurs is typically one-third of the total P sorption capacity. This information, combined with the results of nutrient budgeting and proposed P loadings, can be used to estimate the sustainable life of an effluent application area. This is based on the $kg{\cdot}ha^{-1}$ of P that can be sorbed in the soil before soil P saturation is reached in comparison with the accumulated P loadings and estimated P export in crops and pastures. Note that P sorption and buffering are theoretical parameters, and any interpretation must tempered by operational experience and monitoring of available P and total P levels in both effluent and soils. This process is detailed in Kruger et al. (1995).

Monitoring phosphorus

Details on monitoring P throughout a dairy effluent management system are provided in chapter 7 'Monitoring and sampling'.

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3.4 Potassium

For general management guidelines pertinent to all nutrients, see chapter 3.1 ‘Nutrient budgeting’. Most issues pertinent to K are dealt with in that chapter, but some additional issues specific to K are dealt with here.

Although a significant proportion of Australian soils have adequate natural levels of available K for plant growth, K deficiencies do occur, typically in higher-rainfall areas, on sandy soils and in coastal areas (Gourley 1999). On dairy farms, K typically needs to be applied to soils to maintain adequate levels for crop or pasture maintenance particularly, where hay is exported or grazing is intensive (Hosking 1986).

Dairy effluent contains variable but often significant levels of K (typically as salts), and can be used to supply pasture or crop K requirements (McDonald *et al.* 2005). Significant K loadings may be applied to paddocks in dairy effluent, but although grazing and fodder conservation can export significant amounts of K, soil K levels can still become very high (Kruger *et al.* 1995). Excessive quantities of K in soils can lead to animal health problems, soil nutrient imbalances and detrimental environmental impacts (Wang *et al.* 2004). The K collected through an effluent management system therefore needs to be managed prudently.

Potassium in dairy effluent

Typically, dairy pastures and supplementary feeds contain between 1% and 3% of their total dry matter (DM) as K (Hosking 1986). However, the quantity of K within dairy effluent will vary with location and feed type and quantities (Ebeling *et al.* 2003). Small amounts of K are exported in milk, and the remainder is passed out in excrement. Minimal K is lost throughout collection and conveyance of dairy effluent on the farm, and it can be assumed that most K collected will be available for reuse. The amount of K in dairy effluent storages will vary considerably; see chapters 2.3 ‘Anaerobic, aerobic and facultative ponds’ and 2.8 ‘Desludging and pond closure’ for typical K concentrations in dairy effluent and sludge. Although these chapters can provide a guide, it is more accurate to analyse the dairy effluent in each individual case (Waters 1999).

Adverse potassium impacts

Potassium losses

Excessive quantities of K in soils can lead to K losses off site through leaching into groundwater or surface runoff. Although K is largely retained in soils, it can leach from coarse, sandy, well drained soils, where loadings are excessive or through bypass flow mechanisms (Gourley 1999, Price 2006). K can be removed in surface runoff from rainfall or irrigation, becoming suspended in the water, typically adsorbed to soil particles (McCaskill *et al.* 2003). Higher concentrations of K in surface runoff or leachate are associated with sites with a high use of K fertiliser or effluent application, especially where soil incorporation does not occur (Fleming and Cox 2001, McCaskill *et al.* 2003). To help minimise the risks of K export, careful monitoring of soil and effluent K levels is required, along with responsive management. In addition, surface runoff should flow into drainage lines and recycling ponds so that no contaminated runoff leaves the farm.

Potassium salts

As K can occur in the soil as a salt, high soil K salt contents can contribute to soil salinity, leading to reduced pasture production on highly saline soils (Hosking 1986). Water-soluble K has been implicated in an adverse effect on soil structural stability in a similar way to sodium (Smiles 2006). Smiles (2006) questions the presumption that Australian soils have few structural problems associated with K and suggests that areas with significant levels of K relative to sodium can suffer adverse soil physical consequences resulting from K just as much as from sodium.

Potassium and stock health

Excessive quantities of K in soils can increase the risk of stock health problems, notably calcium deficiency (milk fever or hypocalcaemia) and magnesium deficiency (grass tetany or hypomagnesaemia), typically on grass-dominant pastures (Wang *et al.* 2004). Excessive soil K levels can result in luxury uptake by pasture, thus increasing K intake by animals (Hosking 1986). The high K concentration in pasture suppresses the uptake of calcium and magnesium by stock, leading to low concentrations of each in the cow's bloodstream (Hosking 1986). These stock health disorders can be managed. Dairy cows are most susceptible to high K levels in the diet during the transition period (before calving) and early lactation. Not grazing cows on areas where effluent has been applied during these times, particularly on consecutive days, will minimise the risk of grass tetany. In addition, grazing the pasture when ryegrass has reached the three-leaf stage is recommended, because the concentrations of Ca and Mg will have increased in the plant by that stage. Magnesium oxide can be added to stock feed to reduce the risk of grass tetany. To minimise the risk of grass tetany and milk fever, annual applications of K should not exceed 120 kg·ha^{-1}, and single applications should not exceed 60 kg·ha^{-1} (Gourley 1999, Hosking 1986).

Potassium management

Detailed nutrient budgeting is dealt with in chapter 3.1 'Nutrient budgeting'. Soil analysis for K is a reliable method of assessing soil K requirements. Soil should be analysed before effluent or fertiliser is applied to assist in determination of appropriate K loadings (Gourley 1999, Gourley *et al.* 2007). Gourley *et al.* (2007) found that available K levels for pastures measured by the commonly used Colwell, Skene and exchangeable K soil tests are strongly correlated with one another, and there was no statistical dependence with state, region or cation exchange capacity. The Colwell K test, preferred by Gourley *et al.* (2007), did show significant dependence on soil texture class.

Gourley *et al.* (2007) calculated that the critical soil-available K level to achieve 95% of maximum pasture production was as detailed in Table 1.

Table 1. Critical Colwell K soil test values for four soil texture classes and the equations describing the relationship between Colwell value and percentage of maximum pasture yield (Gourley et al. 2007).

Soil texture	Critical value[1]	Confidence interval[2]	Number of experiments	Equation[3] % maximum yield =
Sand	126	109-142	50	$100 \times (1 - e^{-0.024 \times \text{Colwell K}})$
Sandy loam	139	126-157	122	$100 \times (1 - e^{-0.022 \times \text{Colwell K}})$
Sandy clay loam	143	127-173	75	$100 \times (1 - e^{-0.021 \times \text{Colwell K}})$
Clay loam	161	151-182	194	$100 \times (1 - e^{-0.019 \times \text{Colwell K}})$

1: Soil test value (mg·kg^{-1}) at 95% of predicted maximum pasture yield.
2: 95% chance that this range covers the critical soil test value.
3: e ≈ 2.71828.

3.4 Potassium

The amount of natural K and the amount of K from effluent or fertiliser which is or becomes available for plants depends on a soils' physical and chemical parameters, including particle size, clay mineralogy, moisture status, organic matter and soil pH (Hosking 1986). The amount of available K can vary greatly across a paddock (Gourley 1999).

Monitoring potassium

Details on monitoring K throughout a dairy effluent management system are provided in chapter 7 'Monitoring and sampling'.

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3.5 Trace elements

For general management guidelines pertinent to all nutrients, see chapter 3.1 'Nutrient budgeting'. Trace elements in dairy effluent can reach concentrations that have adverse impacts on the dairy production system or on the environment, and for this reason require careful management. Although trace element deficiencies can occur in dairy production systems, they are related to agronomic management and are therefore not dealt with here.

Trace element and contaminant excess

Care is required in the application of dairy effluent to land, because trace elements (copper, zinc etc.), heavy metals (cadmium, arsenic, chromium, mercury etc.), therapeutic compounds and organic materials from pesticides can occur in dairy effluent (McBride and Spiers 2001, Wang *et al.* 2004). Although most dairy effluent is unlikely to have excess concentrations of these contaminants, an excess build-up can result in the over-application of these to land and a subsequent build-up in the soil. When trace element or contaminant levels in a soil become excessive, there is the potential for impacts on productivity and the environment, and the risk of plant and animal uptake to levels that can pose a threat to the health of stock or humans. Bolan *et al.* (2003) found that in New Zealand, metals, and especially Zn and Cu, in dairy effluent originated from feed or therapeutic treatments, especially from feed additives and growth promoters.

A study by McBride and Spiers (2001) of both liquid and solid dairy manures in New York state, USA, indicated that concentrations of heavy metals such as cadmium, lead and mercury were low and that those of Cu and Zn were elevated. They concluded that although a significant proportion of Cu and Zn could be attributed to feed additives, some could be attributed to contamination of the manure by soil or other wastes (feed, bedding, therapeutics etc).

Although the source is unclear, Anon. (2004) cites data on the composition of manures listing Zn concentrations in 'dairy shed solids' of 100 to 200 $mg{\cdot}kg^{-1}$ and in 'cattle' manure of 80 to 283 $mg{\cdot}kg^{-1}$, and Cu concentrations in 'cattle' manure of 14 to 71 $mg{\cdot}kg^{-1}$. These results compare with the data of McBride and Spiers (2001), who found Zn levels in New York dairy manures of 87 to 488 $mg{\cdot}kg^{-1}$ (average 191 $mg{\cdot}kg^{-1}$ dry weight), and Cu levels of 18 to 1100 $mg{\cdot}kg^{-1}$ (average 139 $mg{\cdot}kg^{-1}$). Although again the source is unclear, the maximum recommended limits for contaminants in animal manures applied to land as cited by Anon. (2004) are listed in Table 1.

Table 1. Maximum recommended limits of contaminants in animal manures applied to land (anon. 2004).

Contaminant	Limit ($mg{\cdot}kg^{-1}$)	Contaminant	Limit ($mg{\cdot}kg^{-1}$)
Arsenic	20	DDT group	0.5
Cadmium	1	Aldrin	0.05
Chromium	400	Dieldrin	0.05
Copper	100	Chlordane	0.05
Lead	150	Heptachlor	0.05
Mercury	1	Hexachlorobenzene	0.05
Nickel	60	Hexachlorocyclohexanes	0.05
Selenium	3	Polychlorinated biphenyls	0.05
Zinc	200		

Upper limits for contaminants in irrigation waters applied to soils, of relevance to dairy liquid effluent ANZECC and ARMCANZ (2000), are listed in Table 2.

3.5 Trace elements

Table 2. Long-term trigger values (LTV), short-term trigger values (STV) and soil cumulative contaminant loading limits (CCL) for heavy metals in agricultural irrigation water (ANZECC & ARMCANZ 2000).

Element	Suggested soil CCL ($kg \cdot ha^{-1}$)	LTV in irrigation water (long-term use—up to 100 y) ($mg \cdot L^{-1}$)	STV in irrigation water (short-term use—up to 20 y) ($mg \cdot L^{-1}$)
Aluminium	ND	5	20
Arsenic	20	0.1	2.0
Beryllium	ND	0.1	0.5
Boron	ND	0.5	
Cadmium	2	0.01	0.05
Chromium	ND	0.1	1
Cobalt	ND	0.05	0.1
Copper	140	0.2	5
Fluoride	ND	1	2
Iron	ND	0.2	10
Lead	260	2	5
Lithium	ND	2.5 (0.075 on citrus)	2.5 (0.075 on citrus)
Manganese	ND	0.2	10
Mercury	2	0.002	0.002
Molybdenum	ND	0.01	0.05
Nickel	85	0.2	2
Selenium	10	0.02	0.05
Uranium	ND	0.01	0.1
Vanadium	ND	0.1	0.5
Zinc	300	2	5

Trigger values should be used only in conjunction with information on each individual element and the potential for off-site transport of contaminants.
ND = not determined; insufficient background data to calculate CCL.

ANZECC & ARMCANZ (2000) provide the following explanation of Table 2:

'The long-term trigger value (LTV) is the maximum concentration ($mg \cdot L^{-1}$) of contaminant in the irrigation water which can be tolerated assuming 100 years of irrigation.

The short-term trigger value (STV) is the maximum concentration ($mg \cdot L^{-1}$) of contaminant in the irrigation water which can be tolerated for a shorter period of time (20 years) assuming the same maximum annual irrigation loading to soil as for LTV.

The LTV and STV values have been developed: (1) to minimise the build-up of contaminants in surface soils during the period of irrigation; and (2) to prevent the direct toxicity of contaminants in irrigation waters to standing crops. Where LTV and STV have been set at the same value, the primary concern is the direct toxicity of irrigation water to the standing crop (e.g. for lithium and citrus crops), rather than a risk of contaminant accumulation in soils and plant uptake.

The trigger value for contaminant concentration in soil is defined as the cumulative contaminant loading limit (CCL). The CCL is the maximum contaminant loading in soil defined in gravimetric units ($kg \cdot ha^{-1}$) and indicates the cumulative amount of contaminant added, above which site-specific risk assessment is recommended if irrigation and contaminant addition is continued.

Once the CCL has been reached, it is recommended that a soil sampling and analysis program be initiated on the irrigated area, and an environmental impact assessment of continued contaminant addition be prepared. As background

concentrations of contaminants in soil may vary with soil type, and contaminant behaviour is dependent on soil texture, pH, salinity, etc., it should be noted that CCLs may be overly protective in some situations and less protective in others. The CCL is designed for use in soils with no known history of contamination from other sources. When it is suspected that the soil is contaminated before commencement of irrigation, background levels of contaminants in the soil should be determined and the CCL adjusted accordingly.

The trigger values assume that irrigation water is applied to soils and that soils may reduce contaminant bioavailability by binding contaminants and reducing concentrations in solution.'

In reference to Cu and Zn levels and by comparison between Table 2 and the data on manure concentrations by Anon. (2004) and McBride and Spiers (2001), the levels of Cu and Zn typically found in dairy manures is considerably higher than ANZECC & ARMCANZ (2000) recommend should be applied in irrigation water. Bolan *et al.* (2003) state that the majority of Cu and Zn in dairy effluent resides in sludge, and that only a small fraction ends up in liquid effluent. However, they found that when both the solid and liquid portions of dairy effluent were applied at a rate to supply typical N requirements of pastures, Cu and Zn were applied at rates tens of times higher than the typical pasture requirements, and that these metals were likely to build up in the soil. McBride and Spiers (2001) in their New York study found that Cu and Zn concentrations in the dairy manures were at levels where, if the manure was applied at rates to supply typical P requirements, Cu and Zn would be applied at rates hundreds of times greater than recommended annual loadings.

Managing trace elements in dairy effluent

Apart from monitoring of dairy effluent trace element and containment levels before land application and adherence to the thresholds listed in Table 2, no guidelines were found for the management of trace elements and containments, especially Cu and Zn, in dairy effluent. Practices that minimise the addition or accumulation of these constituents to dairy effluent in the first place are probably the best course of action, but these may not always be practical. Dilution of effluent may be another option. More research is required to determine thresholds for trace elements and contaminants in dairy effluent that is to applied to land to avoid the development of adverse impacts.

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3.6 Salinity

Salinity is the presence of high levels of soluble salts in soils or waters. Management of salinity is important, as elevated salt levels can have detrimental effects on production and the environment. Elevated levels in dairy effluent are often a potential problem, as salts are ubiquitous in the production system and it is difficult to separate them from effluent. Best practice for the management of water, nutrients and salts in dairy effluent is land application. A range of methods are available to minimise salt loading and to manage land application areas.

Salinity units and conversions

The salinity level of water is typically reported as either electrical conductivity (EC) or total dissolved solids (TDS).

It is critical to note what units a water salinity measurement is reported in. Water EC can be measured in:

- microsiemens per centimetre ($\mu S \cdot cm^{-1}$)
- millisiemens per centimetre ($mS \cdot cm^{-1}$)
- decisiemens per metre ($dS \cdot m^{-1}$).

The TDS in water can be measured in:

- milligrams per litre ($mg \cdot L^{-1}$)
- parts per million (ppm).

Note that 1000 EC units = 1000 $\mu S \cdot cm^{-1}$ = 1.0 $mS \cdot cm^{-1}$ = 1.0 $dS \cdot m^{-1}$ = 640 $mg \cdot L^{-1}$ TDS = 640 ppm TDS.

To convert EC units to TDS: multiply by 0.64.

To convert TDS to EC units: multiply by × 1.5625.

Minimising salinity

Where there is a risk of salinity levels becoming elevated in the dairy effluent stream, it is prudent to minimise salt accumulation in this stream where possible. The following dairy management components should be assessed.

Salt importation

- Washdown water—one of the main determinants of effluent salinity levels is the salinity of water used for washdown, as the proportion of effluent generated by this process can be considerable.
- Water—stock drinking water may contain salts.
- Feed—feeds can also contain salts, especially by-products or supplements.
- Cleaning agents—milking shed sanitisers can often import significant levels of salts. Low-salinity or differing-salinity alternatives are available.

Salt accumulation

Salts can readily accumulate in dairy effluent where the effluent is recycled for washdown. Salts can concentrate in ponds through evaporation.

Storage risks

It is important to ensure that effluent storage salinity levels do not become excessive, as this can impede pond biological interactions (see chapter 2.3 'Anaerobic, aerobic and facultative ponds').

Applying effluent to land

Salinity risks

Care is required where dairy effluent with elevated salinity is applied to land, as salinity levels can influence production (e.g. pasture growth) and the environment. Table 1 indicates the risks associated with the application of water with elevated salt levels. Elevated salt loadings can result in an accumulation of salts in the pasture or crop rootzone that can affect yield and therefore water and nutrient use. In addition, the mobilisation of excess salts can have adverse off-site environmental impacts.

Table 1. Salinity classes of irrigation waters (Environment Protection Authority 1991)**.**

Class	TDS* (mg·L^{-1})	EC* (µS·cm^{-1})	Comments
1	0–175	0–270	Can be used for most crops on most soils by all methods or water application with little likelihood that a salinity problem will develop. Some leaching is required, but this will occur under normal irrigation practices, except in soils of extremely low soil permeabilities.
2	175–500	270–780	Can be used if a moderate amount of leaching occurs. Plants with moderate salt tolerance can be grown, usually without special salinity management practices. Sprinkler irrigation with the more saline waters in this class may cause leaf scorch on salt-sensitive crops.
3	500–1500	780–2340	Do not use the more saline waters in this class on soils with restricted drainage. Even with adequate drainage, best practice management controls for salinity may be required, and the salt tolerance of the plants to be irrigated must be considered.
4	1500–3500	2340–5470	For use, soils must be permeable with adequate drainage. Water must be applied in excess to provide considerable leaching, and salt-tolerant crops should be grown.
5	>3500	>5470	Not suitable for irrigation except on well drained soils under good management, especially leaching. Restrict to salt-tolerant crops, or for occasional emergency use.

* See conversions at end of this chapter.

An indication of typical salinity levels found in dairy effluent storages is provided by (Waters 1999), who quotes levels from south-western Victoria measured in 1996 ranging from 2800 to 7700 EC units (µS·cm^{-1}).

Assessing application impacts

In assessing the risks associated with the application to land of effluent with elevated salinity levels, assess the following issues:

- The existing salinity status of the soils and the surface and groundwater in the region and on the farm.
- The potential salinity level on application, allowing for any dilution (refer to Table 1).
- The proposed method of effluent application (spray application of saline waters can cause leaf burn in certain crops or pastures depending on salinity levels, crop susceptibility, application timing and temperature).
- Climatic variables, especially rainfall–evaporation interactions (levels of water application to sustain crop or pasture growth, leaching attained from applied water or from rainfall).

An assessment of these factors will help determine whether there is a significant risk of adverse impacts on production or the environment resulting from salinity applied in effluent. In cases where Class 3, 4 or 5 water is to be applied (Table 1), where significant salinity effects exist in the region or on the farm, where site variables (including adverse drainage, or soil or groundwater conditions) indicate a risk, or where climatic variables indicate a risk of salt accumulation, use salinity budgeting to assess the risks.

The salinity status of the potential effluent reuse site must be considered in relation to several factors:

- Existing soil salinity levels are often variable and are of more value when recorded over time and assessed in conjunction with water application, rainfall, surrounding surface or groundwater depth and salinity level fluctuations.
- Groundwater depth and salinity levels: Shallow (typically <2.0 m below natural surface) and saline groundwaters can make site salinity management difficult.
- Groundwater beneficial uses and environmental interactions: Increased salinity levels can significantly affect other users. Impacts can result where groundwaters approach or broach the surface or interact with surface waters.
- Site surface drainage: Good surface drainage is preferable.
- Soil permeability: Soils with low permeability can make site salinity management difficult. In clayey soils, the impacts of irrigation, rainfall, salt and sodium on soil permeability need to taken into account.
- Soil profile potential leaching fraction: If the amount of leaching that can be achieved is low, salinity management will be more difficult.

Salinity budgeting

Where elevated salinity loadings to land are sustained or where there is significant risk of adverse effects on production and the environment resulting from salinity (see above), use salinity to budgeting assess the suitability of land application by considering a number of interactive factors. A number of salinity budgeting options are available (ANZECC & ARMCANZ 2000, Ayers and Westcot 1989). These typically calculate the estimated soil salinity level from applied water quality, any dilution, application rate, rainfall and the achievable leaching fraction. The predicted soil salinity is correlated with soil salinity tolerances of crops (Ayers and Westcot 1989), thereby providing an estimate of yield loss resulting from salinity for a particular plant species. Salinity budgeting can be used to indicate a leaching requirement that will maintain sufficiently low soil salinity to limit production losses to an acceptable level. A comparison between this leaching requirement and the perceived achievable leaching fraction provides one component of an assessment of dairy effluent application suitability.

Leaching of salts

Where there is significant risk of adverse production and environmental impacts from salinity (see above), some leaching of salts is typically required to maintain sufficiently low rootzone salinity levels. In addition to the site production aspects assessed through salinity budgeting, the fate of these leached salts must also be taken into consideration. The estimation of achievable leaching fraction is typically based on experience and needs to consider any soil drainage enhancement, soil physical characteristics, and the interactions of sodium, calcium and salinity with soil structure (Doorenbos and Pruitt 1984). The interactions of these factors dictate the changeable nature of leaching fractions and require that a range of leaching fractions be considered. Use rainfall data and water budgeting in the salinity budgeting. Typically leaching should be facilitated by rainfall. Where this is insufficient, additional water over and above plant water demands should be applied to facilitate leaching (Environment Protection Authority 1991). Monitoring of topsoil and subsoil salinity is typically recommended to ascertain the effectiveness of leaching (see chapter 7 ‘Monitoring and sampling’).

Salinity budgeting process

Although a range of salinity assessment options are available, one process for assessment of the suitability of land application, sourced and adapted from ANZECC & ARMCANZ (2000), details five key steps:

Step 1

Identify the key interactive factors, including:

- salinity levels of waters to be applied
- shandying ratio and water application rates
- the resulting salt loadings
- climatic information, including rainfall and evaporation
- soil properties and relevant hydrological and hydrogeological features
- plant salt tolerances
- relevant site management.

Step 2

Estimate the achievable leaching fraction under the proposed water application regime (see ANZECC & ARMCANZ (2000)).

Step 3

Estimate the new average root zone salinity (see ANZECC & ARMCANZ (2000)). Average rootzone salinity is the key limitation to plant growth in response to the application of water with elevated salinity. However, poor soil structure resulting from salinity and sodium can also reduce plant yields by limiting aeration, water infiltration and root growth.

Step 4

Estimate relative plant yield loss due to salinity (see ANZECC & ARMCANZ (2000)).

Step 5

Consider salinity impacts within the broader catchment, such as regional water tables, groundwater pollution and surface water quality.

Steps 2 to 4 cover what is typically considered salinity budgeting. Further salinity budgeting examples are provided at the end of the chapter. Software such as SALF and SALF PREDICT can estimate the parameters necessary for a detailed assessment of irrigation water quality in relation to soil properties, rainfall, water quality and plant

salt tolerance. This type of software is based on summer rainfall areas and should be used with some caution in winter rainfall areas (DPI 2001).

Salinity management and monitoring

Where there is significant risk of adverse effects on production and the environment from salinity, success is often determined by a combination of prudent site management and careful monitoring. The actual yield reduction due to salinity will vary considerably and will depend on a range of management factors, but will be dictated mainly by irrigation water salinity levels and the degree of leaching attained with rainfall and irrigation water. The following salinity management strategies can reduce the risk of adverse impacts:

- Good site design, the provision of good surface drainage and accurate water application infrastructure (such as sprinklers).
- Proficient water application scheduling to match plant demands and leaching requirements, in conjunction with regular climatic and soil moisture monitoring (see chapter 7 'Monitoring and sampling').
- The regular application of gypsum according soil monitoring results.
- Where salinity budgeting indicates a risk, monitor management practices, soils, crop production levels, groundwater and surface runoff. Assess monitoring results in conjunction with the results from the monitoring of all site environmental, management and production parameters.
- Where salinity risks are suspected, a hand-held electrical conductivity meter can be a valuable and relatively cheap tool providing for easy field assessment of water (e.g. water source, effluent pond), the effluent application site (e.g. paddock runoff), the farm (e.g. runoff recycling dam, groundwater) and within the broader catchment (e.g. neighbouring creeks).
- Risks can be reduced by spreading small amounts of effluent over a large area (Waters 1999).

Example salinity budget calculations

Example 1—Leaching requirements for perennial pasture

What leaching fraction is required and what will the yield loss be, if any, if saline bore water (2000 EC units = 2000 $\mu S \cdot cm^{-1}$ = 2.0 $dS \cdot m^{-1}$ = 1280 ppm TDS) is applied to a perennial ryegrass–clover pasture (*Lolium perenne* and *Trifolium repens*) where the bore water is shandied in a ratio of 1:1 with irrigation water at 300 EC units (= 300 $\mu S \cdot cm^{-1}$ = 0.3 $dS \cdot m^{-1}$ = 192 ppm TDS) and effective leaching winter rainfall (Ayers and Westcot 1989)?

Bore water salinity	2000 EC units = 2.0 $dS \cdot m^{-1}$
Irrigation water	300 EC units = 0.3 $dS \cdot m^{-1}$
Dilution ratio	1:1
Applied water salinity	1150 EC units = 1.15 $dS \cdot m^{-1}$
Irrigation application rate	7.0 $ML \cdot ha^{-1}$*
Average irrigation season rainfall	300 mm*
Estimated rainfall salinity	50 $dS \cdot m^{-1}$
Average irrigation + rainfall salinity	0.8 $dS \cdot m^{-1}$

* Values typically derived from water budgeting—see chapter 3.9 'Hydraulic application rate and scheduling'.

Soil salinity thresholds and yield reductions (Ayers and Westcot (1989):

Perennial ryegrass:	No yield reduction = 5.6 dS·m^{-1}
	10% yield reduction = 6.9 dS·m^{-1}
	25% yield reduction = 8.9 dS·m^{-1}
White clover:	No yield reduction = 1.5 dS·m^{-1}
	10% yield reduction = 2.3 dS·m^{-1}
	25% yield reduction = 3.6 dS·m^{-1}

Calculate leaching requirement as (irrigation water salinity) / (5 × soil salinity threshold – irrigation water salinity) (Ayers and Westcot 1989):

Ryegrass:	0.8 / (5 × 5.6 – 0.8) × 100% ≈ 3%
Clover:	0.8 / (5 × 1.5 – 0.8) × 100% ≈ 12%

Additional water required for leaching to achieve no yield reduction:

Ryegrass leaching requirement of 3%	0.21 ML (i.e. 3% of 7 ML·ha^{-1} annual irrigation application)
Clover leaching requirement of 12%	0.84 ML

This calculation indicates that a 3% leaching fraction will be required to avoid yield loss in perennial ryegrass, and 12% for white clover. If we assume that the soil could accommodate a leaching fraction of 12%, an additional 0.84 ML of irrigation water would need to be applied over and above plant water requirements to sufficiently leach salts from the pasture rootzone for white clover.

Let's say that an examination of the soil profile on this dairy farm indicates an achievable leaching fraction of 5%. Example 2 below shows how calculate the yield loss from this water application scenario.

Example 2—Estimated yield reduction at set leaching fraction

What will the yield loss be, if any, if irrigation water diluted to 1150 EC units (= 1150 μS·cm^{-1} = 1.15 dS·m^{-1} = 736 ppm TDS) is applied to a perennial ryegrass–clover pasture where the achievable leaching fraction is 5% (Ayers and Westcot 1989)?

Applied water salinity	1150 EC units = 1.15 dS·m^{-1}
Irrigation application rate	7.0 ML·ha^{-1}*
Average irrigation season rainfall	300 mm*

* Values typically derived from water budgeting—see chapter 3.9 'Hydraulic application rate and scheduling'.

Estimated rainfall salinity	50 dS·m^{-1}
Average irrigation + rainfall salinity	0.8 dS·m^{-1}
Achievable leaching fraction	5%

Calculate resulting soil salinity as (irrigation water salinity / leaching fraction) + irrigation water salinity) divided by 5 (Ayers and Westcot 1989):

Resulting soil salinity	(0.8/0.05 + 0.8) / 5 = 3.36 dS·m^{-1}

The application of water with a salinity level of 1150 EC units at 7.0 ML·ha^{-1}, under average annual rainfall and a leaching fraction of 5%, will give the soil a salinity level of 3.36 dS·m^{-1}.

3.6 Salinity

Soil salinity thresholds and yield reductions (Ayers and Westcot 1989):

Perennial ryegrass:	No yield reduction = 5.6 $dS{\cdot}m^{-1}$
	10% yield reduction = 6.9 $dS{\cdot}m^{-1}$
	25% yield reduction = 8.9 $dS{\cdot}m^{-1}$
White clover:	No yield reduction = 1.5 $dS{\cdot}m^{-1}$
	10% yield reduction = 2.3 $dS{\cdot}m^{-1}$
	25% yield reduction = 3.6 $dS{\cdot}m^{-1}$

A soil salinity level of 3.36 $dS{\cdot}m^{-1}$ will not cause any yield reduction in perennial ryegrass, as the soil salinity threshold for yield reduction in ryegrass is 5.6 $dS{\cdot}m^{-1}$. However, in the more salt-sensitive white clover, it will cause a predicted yield reduction of 23% (by interpolation).

The managers of the example dairy farm need to decide whether they can tolerate a 23% loss in white clover production and, if not, will have to reduce the amount of salt applied through, for example, further dilution with the bore water.

The impacts of leaching these salts with irrigation water and winter rainfall need to be taken into consideration, especially in relation to other areas of the farm and in the broader catchment, notably regional water table depth, groundwater quality and surface water impacts.

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3.7 Sodicity

Sodicity is the presence of a high proportion of sodium ions relative to calcium plus magnesium ions in water or to the cation exchange capacity of a soil.

The potential for sodium to degrade soil structure, resulting in erosion, reduced permeability and subsequent waterlogging, and a decline in plant growth, in part determines the suitability of dairy effluent for land application. The sodium levels of dairy effluent and existing and potential soil sodicity levels can be used to calculate the risk associated with land application.

Elevated sodium levels in dairy effluent and the subsequent elevated soil sodicity levels can often be managed through a range of strategies, such as the application of gypsum or organic matter.

Sodium in dairy effluent

In relation to sodium, the suitability of water for application to land should be evaluated on the basis of a range of criteria that indicate the potential of the water to harm plant growth or to create soil conditions hazardous to plant growth or to animals or humans in contact with the plants or soil (ANZECC & ARMCANZ 2000). Dairy effluent sodicity levels are assessed through calculation of the sodium adsorption ratio (SAR), which is a measure of the amount of sodium present in the effluent relative to calcium plus magnesium (Rengasamy and Olsson 1993).

Calculation of SAR

The SAR is calculated with Equation 1; note that Na^{+}, Ca^{2+} and Mg^{2+} are measured in $cmol^{+}\cdot kg^{-1}$. For this calculation the dairy effluent must first be analysed for all three ions.

$$SAR = Na^{+} / \sqrt{((Ca^{2+} + Mg^{2+}) / 2)} \qquad (1)$$

The SAR can be adjusted (SAR_{adj}) by Equation 2 to take into account the effects of electrical conductivity (EC), carbonate and bicarbonate (Environment Protection Authority 1991). For this, Ca^{2+} is replaced by Ca_x, which is determined by the EC of the effluent and the bicarbonate-to-calcium ratio (HCO_3^{-}/Ca^{2+}) of the effluent (Table 2) (Environment Protection Authority 1991). Concentrations of Na^{+} and Mg^{2+} in milliequivalents per litre ($mEq\cdot L^{-1}$) are determined from Table 1.

Table 1. Factors for conversion of ions from $mg\cdot L^{-1}$ to $mEq\cdot L^{-1}$ (Environment Protection Authority 1991).

		Conversion	
Na^{+}	$mg\cdot L^{-1}$	× 0·0435	$mEq\cdot L^{-1}$
Mg^{2+}	$mg\cdot L^{-1}$	× 0·0833	$mEq\cdot L^{-1}$
Ca^{2+}	$mg\cdot L^{-1}$	× 0·0500	$mEq\cdot L^{-1}$
HCO_3^{-}	$mg\cdot L^{-1}$	× 0·0164	$mEq\cdot L^{-1}$
$CaCO_3$	$mg\cdot L^{-1}$	× 0·0200	$mEq\cdot L^{-1}$

$$SAR_{adj} = Na^{+} / \sqrt{((Ca_x + Mg^{2+}) / 2)} \qquad (2)$$

3.7 Sodicity

Table 2. Values of Ca_x determined by salinity (EC) and the bicarbonate-to-calcium ratio (HCO_3^-/Ca^{2+}) of the effluent—for use in Equation 2 (Environment Protection Authority 1991).

Ratio of HCO_3/Ca	Irrigation water EC (dS m^{-1})											
	0.10	0.20	0.30	0.50	0.70	1.00	1.50	2.00	3.00	4.00	6.00	8.00
0.05	13.20	13.61	13.92	14.40	14.79	15.26	15.91	16.43	17.28	17.97	19.07	19.94
0.10	8.31	8.57	8.77	9.07	9.31	9.62	10.02	10.35	10.89	11.32	12.01	12.56
0.15	6.34	6.54	6.69	6.92	7.11	7.34	7.65	7.90	8.31	8.64	9.17	9.58
0.20	5.24	5.40	5.52	5.71	5.87	6.06	6.31	6.52	6.86	7.13	7.57	7.91
0.25	4.51	4.65	4.76	4.92	5.06	5.22	5.44	5.62	5.91	6.15	6.52	6.82
0.30	4.00	4.12	4.21	4.36	4.48	4.62	4.82	4.98	5.24	5.44	5.77	6.04
0.35	3.61	3.72	3.80	3.94	4.04	4.17	4.35	4.49	4.72	4.91	5.21	5.45
0.40	3.30	3.40	3.48	3.60	3.70	3.82	3.98	4.11	4.32	4.49	4.77	4.98
0.45	3.05	3.14	3.22	3.33	3.42	3.53	3.68	3.80	4.00	4.15	4.41	4.61
0.50	2.84	2.93	3.00	3.10	3.19	3.29	3.43	3.54	3.72	3.87	4.11	4.30
0.75	2.17	2.24	2.29	2.37	2.43	2.51	2.62	2.70	2.84	2.95	3.14	3.28
1.00	1.79	1.85	1.89	1.96	2.01	2.09	2.16	2.23	2.35	2.44	2.59	2.71
1.25	1.54	1.59	1.63	1.68	1.73	1.78	1.86	1.92	2.02	2.10	2.23	2.33
1.50	1.37	1.41	1.44	1.49	1.53	1.58	1.65	1.70	1.79	1.86	1.97	2.07
1.75	1.23	1.27	1.30	1.35	1.38	1.43	1.49	1.54	1.62	1.68	1.78	1.86
2.00	1.13	1.16	1.19	1.23	1.26	1.31	1.36	1.40	1.48	1.54	1.63	1.70
2.25	1.04	1.08	1.10	1.14	1.17	1.21	1.26	1.30	1.37	1.42	1.51	1.58
2.50	0.97	1.00	1.02	1.06	1.09	1.12	1.17	1.21	1.27	1.32	1.40	1.47
3.00	0.85	0.89	0.91	0.94	0.96	1.00	1.04	1.07	1.13	1.17	1.24	1.30
3.50	0.78	0.80	0.82	0.85	0.87	0.90	0.94	0.97	1.02	1.06	1.12	1.17
4.00	0.71	0.73	0.75	0.78	0.80	0.82	0.86	0.88	0.93	0.97	1.03	1.07
4.50	0.66	0.68	0.69	0.72	0.74	0.76	0.79	0.82	0.86	0.90	0.95	0.99
5.00	0.61	0.63	0.65	0.67	0.69	0.71	0.74	0.76	0.80	0.83	0.88	0.93
7.00	0.03	0.50	0.52	0.53	0.55	0.57	0.59	0.61	0.64	0.67	0.71	0.74
10.00	0.39	0.40	0.41	0.42	0.43	0.45	0.47	0.48	0.51	0.53	0.56	0.58
20.00	0.24	0.25	0.26	0.26	0.27	0.28	0.29	0.30	0.32	0.33	0.35	0.37

Consequences of SAR

The SAR of dairy effluent can vary considerably and depends on the sodium levels in the water used in the dairy, on dairy shed and feedpad management and on effluent treatment. Typical and usually safe SAR levels for dairy effluent are 1 to 6, but levels up to 10 are not uncommon. The SAR can approach 20 in some cases, and values over 20 indicate severe sodicity problems.

In assessing the suitability of effluent for land application, you must also assess the propensity for sodium-related problems to develop. Some soils, especially those with a high clay content, poor subsoil structure or low permeability, can retain excessive exchangeable sodium, which breaks down soil structure and reduces permeability under rainfall or freshwater irrigation (Rengasamy *et al.* 1984). Figure 1 indicates the risk of permeability problems developing. However, the interaction between soil structure, exchangeable sodium and salinity is complex, and the permeability of the soil resulting from these interactions readily fluctuates, especially over the course of a season (Rengasamy and Olsson 1991, Rengasamy and Olsson 1993).

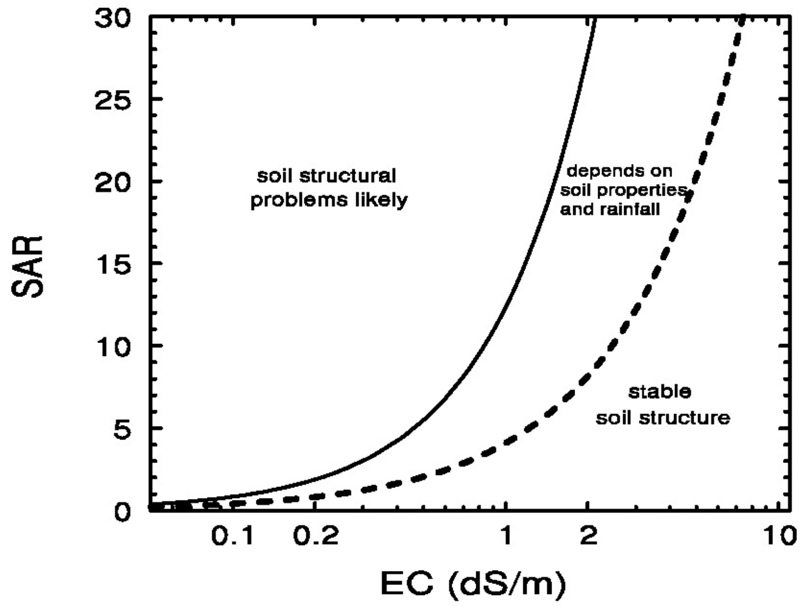

Figure 1. The risk of soil structural change in relation to the salt content of irrigation waters (ANZECC & ARMCANZ 2000).

Sodium in soils

Sodium at high concentrations in soils, especially at a high ratio to other cations, has a detrimental effect on soil structure and thus on plant growth (Rengasamy *et al.* 1984). Sodicity degrades soil structure by breaking down clay aggregates, which makes the soil more susceptible to erosion and dispersion (Rengasamy and Olsson 1991). The dispersed soil particles are then washed away by low-salinity water such as rainfall or irrigation (Rengasamy and Olsson 1993).

Calculating soil exchangeable sodium percentage

Where high soil sodium levels are likely (e.g. soils with high clay content, poor subsoil structure or low permeability), it is important to regularly assess soil exchangeable sodium percentage (ESP) levels to gauge soil structural stability.

The ESP considers the balance between sodium ions and the cation exchange capacity (CEC) of a soil. It is calculated by dividing exchangeable sodium by the CEC or, as the majority of the CEC is due to sodium, potassium, calcium and magnesium, the sum of these four exchangeable cations:

ESP = exch. Na ($cmol^+ \cdot kg^{-1}$) × 100 / (exch. Na + exch. Mg + exch. Ca + exch. K) ($cmol^+$ or $cmol^{2+} \cdot kg^{-1}$)

Table 3 indicates the effects of the soil ESP.

Table 3. Suitability of soil by ESP.

ESP	Rating	Comments
<5	Satisfactory	Insufficient proportion of Na to cause dispersion
5–6	Marginal	Potentially sufficient Na to cause dispersion
6–15	Poor	Likely structural problems caused by high proportion of Na
>15	Very poor	Definite structural problems caused by high proportion of Na

In Australia, a soil with an ESP > 6 is technically termed a sodic soil, and soils with an ESP > 15 are considered to be highly sodic.

Effects of soil sodicity on soil

High concentrations of sodium in a soil create a state of easy dispersion, leading to poor soil physical conditions (Rengasamy and Olsson 1993) such as:

- low hydraulic conductivity, conceivably due to blockage of pores by dispersed colloids
- the downward movement of dispersed material, leading to the formation of a clay pan, which can limit root development and drainage
- unfavourable soil consistency: hard when dry and plastic-sticky when moist; such soils are difficult to work
- a low resistance to slaking, easily leading to the formation of surface crusts, which hamper water infiltration and plant emergence
- waterlogging resulting from the general deterioration of soil drainage associated with the above effects.

Take care in relating poor drainage in the soil to high ESP levels since, although high ESP frequently causes poor drainage, inherent poor drainage characteristics of the soil may also lead to high soil salinity and high ESP values (Rengasamy and Olsson 1993).

Effects of soil sodicity on plant growth

The mainly osmotic effects of salts on crop transpiration and growth are related to total salt concentration rather than to the individual concentrations of specific ions such as sodium. However, sodium typically plays a significant role in salinity, as it is a constituent of the harmful salt sodium chloride. The effects of salts on plants are generally evidenced as reduced transpiration and retarded growth, producing smaller plants with fewer and smaller leaves. Effects of high soil sodicity on plants growth can also materialise in the form of a toxicity or a nutritional imbalance. The effects of specific solutes, or their proportions, especially chloride, sodium and boron, can reduce plant growth. These effects are generally evidenced by leaf burn and defoliation.

Managing sodicity

In applying sodic waters to land it is important to ensure a high level of irrigation design, accurate irrigation infrastructure, proficient irrigation scheduling to match plant demands

and regular detailed monitoring. Strategies for the management of sodicity are similar to those for managing salinity (Rengasamy and Olsson 1991):

- Minimise sodium levels in any applied water wherever possible. This is likely to include sourcing low-sodium dairy shed water and minimising sodium in feedstuffs and sanitisers.
- Regularly apply gypsum at rates determined from the results of regular soil chemistry analyses. Applications typically vary from 1 to 5 $t{\cdot}ha^{-1}$ every 1 to 5 years.
- Provide good surface drainage to divert excess irrigation and rainfall runoff.
- Grow crops that are more tolerant of elevated soil sodium levels.
- Raise soil organic levels.

Monitoring sodicity

Details on monitoring sodium levels are provided in chapter 7 'Monitoring and sampling'.

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3.8 Effluent distribution and irrigation systems

The uniformity of land application (or distribution) of effluent depends on the type of system used to transport and deliver dairy effluent to the land and on farmer expertise, capital cost, labour availability, system complexity, climate, weather, landscape, soil type and the crop or pasture being grown. In irrigation areas (particularly flood irrigation areas), the shandying of effluent with other water sources via conventional irrigation systems is typical. Direct application from a yard or pond is also practised using equipment suited to a higher solids content (see also chapter 2.2 ‘Direct application systems’).

Managing effluent for irrigation

Recycling irrigation tailwater

In some dairy production areas, especially in irrigation areas, irrigation tailwater recycling sumps are common. It is usually recommended that where effluent is distributed via flood irrigation, all farm drainage be contained for reuse on the farm to avoid water degradation from runoff from a site receiving applied effluent (DairyCatch 2006, DPI 2004, McDonald 2002). This runoff could arise directly from the effluent application or indirectly from rainfall shortly after application.

Solids

Dairy effluent must undergo mechanical or gravity separation before it is conveyed through conventional irrigation systems. The solids in dairy effluent comprise nutrients, organic matter and inert materials like gravel, sand and clay which can compromise the performance of conduits and nozzles. Effluent with up to 5% solids can generally be handled as a liquid, and effluent with more than 20% solids can generally be handled as a solid (see chapter 1.1 ‘Physical, biological and chemical components of effluent and manure’). The application of harvested solids to land is discussed in chapter 3.10 ‘Land application of manure and pond sludge’.

Systems for land application

The types of systems available for the application of effluent and irrigation water are many and varied. They include flood irrigation systems, which encompass border-check irrigation, furrow, contour bank, contour ditch and paddy. Other systems include solid set sprinklers, lateral-move travellers, centre-pivot sprinklers, boom sprinklers, bike moves, pop-up sprinklers, pipe and risers, gated pipes, and drip and micro-sprinkles. All conventional irrigation systems can be used to apply dairy effluent to land. However, although effluent can be transported and applied through dedicated pipe and sprinkler systems, these systems are at strong risk of clogging and corrosion. The utility of any system depends on the concentration of nutrients and salts and on the amount of solids, which dictates downslope surface flow prospects and the clogging potential of pumps, pipes and nozzles.

Flood irrigation

Flood irrigation applies water by gravity to moderately sloping land. Slope is important, as effluent must pass across a paddock in 6–10 h to reduce waterlogging; if the slope is too little, excessive infiltration losses will occur, whereas if it is too great, the soil profile will not be adequately wetted. Flood irrigation systems are commonly used for the application of dairy effluent. Effluent which has undergone primary treatment will

distribute more uniformly across the paddock when mixed with water. Soils generally become saturated to the depth of the pasture root zone or below after flood irrigation. As long as evapotranspiration rates are high and profile drainage is possible, the period of saturation is small. These types of systems are not really impeded by solids in the effluent as long as the solids are degradable.

Advantages of flood irrigation:

- low energy use
- minimal land-forming required
- low cost
- low labour.

Disadvantages of flood irrigation:

- unsuitable on undulating or hilly land
- unsuitable on sandy soils
- low application efficiency
- wasteful of water.

The major risks of the more rudimentary flood irrigation systems are the potential for non-uniform distribution of water, nutrients and salt and the exposure to rainfall-induced runoff during and after application. The non-uniformity derives mainly from a poor match between the rate of flow applied at the top of the irrigated area, area slope, area width and soil type. Generally tailwater recycling systems are an essential component. Although these types of systems are often much maligned, they are suited to many soil types and are often the cheapest method of land application both in system establishment and during system operation. Further information on distributing effluent by flood irrigation is documented in McDonald (2002).

Border-check irrigation

Border-check irrigation (BCI) is a form of flood irrigation which allows land with minor slopes to be flood-irrigated by gravity and the tailwater to be captured and reused.

BCI is the most desirable low-energy method for surface-irrigating crops with dairy effluent where topographical and soil conditions are favourable or can be made favourable. Ideal crops for use with BCI systems include lucerne, annual and perennial clover–ryegrass pasture, other deep-rooted pasture types, close-growing timber plantations, forage or cereal crops and orchards.

Land slopes of more than 0.2% but <1% are most ideal for BCI (Wrigley 2002). Grade changes should be slight, and reverse grades must be avoided. Bays should be formed to provide uniform downslope gradient without cross-fall. If irrigation is required on flat land, it is desirable to establish slopes of more than 0.5% through land-forming. Cross-fall is permissible when confined to differences in elevation of 6 to 9 cm between border strips (Wrigley 2002). The hydraulic application efficiency of BCI is generally quoted as 45% to 60%, excluding tailwater reuse. The inputs and outputs of BCI systems can be controlled and automated.

Dairy effluent can be conveyed through a BCI system by being mixed with irrigation water. Effluent should not be conveyed in channels supplying water for stock or domestic purposes. Shandying of effluent with fresh irrigation water is a common way to lower the levels of salts and nutrients being applied and to achieve as high a flow rate as possible to uniformly wet the root zone to an acceptable depth. Environmental control is highly important, to make effective reuse of nutrients so as to meet soil and plant crop requirements. BCI is most suited to duplex clay soils with low permeability. Sandy and loamy soils are generally unsuitable for BCI owing to their high infiltration rates.

Advantages of BCI:

- low energy requirement due to gravity conveyance of effluent
- simple in design
- suited to broadacre crops
- no problems with clogging
- no aerosols or wind drift
- cheap
- rapid reduction in pathogens through exposure
- easy to maintain
- for the Goulburn Murray Irrigation District and Macalister Irrigation District exploits existing infrastructure.

Disadvantages of BCI:

- weed infestation and sediment clogging of channels and drains
- can only be used with shandied irrigation water during the irrigation season
- pumping costs to extract water from a recycling sump
- low application efficiency; high evapotranspiration and percolation
- high cost of land-forming to set up, and associated soil disturbance
- potential crusting of the surface
- possible odours through overloading and non-uniform application.

Furrow irrigation

Furrow irrigation is a method of irrigating the root zone of a plant without the need for water to penetrate vertically through the soil. It allows water to be flooded between ridges, graded furrows or corrugations. The water application efficiency of furrow irrigation is generally <65%. Although the actual grade in the direction of irrigation is typically 0.5% to 1.5%, no land-forming is required beyond filling gullies and removing abrupt ridges. Flow rates depend on the size of furrows and are typically dictated by siphons or bank cuts. Like other flood irrigation systems, furrow irrigation is most suited to clay soils where the potential for leaching is far less, but it is unsuitable for cracking clays.

Furrow irrigation systems are more suited to row crops and fruit production rather than to pastures. They are discouraged where salt levels in soil or effluent are likely to be elevated, as salts can be concentrated in the root zones. Despite these drawbacks, dairy effluent has been applied successfully by furrow irrigation. Effluent is usually siphoned from an irrigation channel and shifted along a furrow under gravity. The effluent permeates vertically and horizontally through the soil and concentrates at or below the root zone of the crop. By moving laterally it can reach the soil surface through capillary action (Wrigley 2002). Weed control and clogging of furrows are problems, especially if raw effluent is applied without being shandied. The system also has a higher risk of human exposure to effluent owing to the high labour requirement. The advantages and disadvantages of furrow irrigation are similar to those of flood and border check irrigation.

Advantages of furrow irrigation:

- low energy use
- water is in close contact with plant root zone
- easily formed.

Disadvantages of furrow irrigation:

- increased labour required
- greater volumes of water required to pass down and over land
- cost of land-forming
- energy required to shift effluent from lagoons to irrigation channels
- cannot be used on steep land.

Sprinkler irrigation

Sprinkler irrigation emulates rainfall and permits the distribution of solids. It usually competes with flood irrigation for broadacre applications, but allows greater control over the quantity applied. It can suit most types of crops and soil types but, unlike flood irrigation, relies on proprietary systems and the performance claims of component suppliers and manufacturers. There are various types of sprinkler systems, including:

- solid-set sprinklers
- linear-move sprinklers
- centre-pivot systems and lateral booms
- rotating booms
- hydrants and bike-move sprinklers.

Sprinkler irrigation of dairy effluent can be an efficient method of land application as long as clogging of jets is avoided. Sprinklers are suitable only if 5% or less of the effluent is solid material (Wrigley 1994). Research in New Zealand indicates that uneven application can result in adverse effects (Houlbrooke *et al.* 2004). Given the potential for particulate clogging of pressure irrigation systems, the selection of a system must be governed by the type of effluent. For raw dairy effluent, large-orifice emitters must be used, and the sprinklers must be dedicated to effluent. Some products on the market overcome the risk of particulate clogging by using flexible nozzles that enable solids to pass through. Under these conditions nozzles need regular replacement, although all sprinkler system warrant regular sprinkler head maintenance and refurbishment.

Flow rates and application rates for sprinkler irrigation need to be lower than soil infiltration rates (see chapter 3.9 ‘Hydraulic application rate and scheduling’). The range of sprinkler systems in use is many and varied, and unfortunately there is very little research to compare them. In addition, much reliance is placed on manufacturers’ claims, which are assessed only in the event of dispute.

Advantages of sprinkler irrigation:

- can be used on rolling terrain (slopes up to 35% depending on equipment and application rate)
- good for high-rainfall areas, where only a small supplementary water supply is needed
- can be used on easily erodible soils with shallow topsoil
- can be used on highly permeable soils, such as sands and loams
- better control of the system allows for more light, frequent applications, even during wet periods, provided the risk of runoff is low
- minimal tailwater
- application efficiency of 60% to 70%, but can be higher with centre pivots
- the rate of application can be adjusted so that surface ponding is avoided

- easy to automate.

Disadvantages of sprinkler irrigation:

- high capital cost
- high operation and maintenance costs
- distribution is subject to wind distortion, although centre pivots are less susceptible
- potential for increases in humidity levels and thus disease
- irrigation with saline effluent can cause damage to plant leaves
- high energy use
- corrosion of components.

Because sprinkler irrigation relies on mechanical devices, these advantages and disadvantages are of a general nature only. Experience indicates that the performance of a new system is incomparable to that of an old, poorly maintained system, and therefore reliance must be placed on experience rather than on documented case studies, which are conspicuous by their absence.

Subsurface irrigation of pastures

There is a significant move towards drip and micro-sprinklers systems, which can provide water to individual plants from a dripper under automatic control. In subsurface irrigation, water is applied beneath the root zone via deep surface channels, pipes or drip tapes. Subsurface irrigation delivers water by capillary action and reduces both evaporation and deep percolation losses. Moisture and nutrients are supplied direct to plants with minimal contact with humans and animals. Subsurface irrigation systems can be used with most soil types and in rolling terrain. The application efficiency is between 80% and 95%.

The application of dairy effluent through subsurface drip irrigation (SSDI) depends on the level of solids in the effluent. Effluent must go through mechanical separation and filtration before being delivered via this system to minimise the likelihood of blocking outlets. Typically a significant volume of irrigation water is necessary for shandying. Unfortunately, the risk of clogging limits prospects for SSDI unless very high standards of filtration are achieved. The size of suspended particulate matter should not exceed 100 µm; clogging is inevitable once particles exceed 300 µm, and filtration is essential. Given this size limitation, SSDI should be used only with highly polished, filtered effluent.

Research is currently under way to determine the potential of SSDI for grazed dairy pastures, but currently drip irrigation is restricted to non-grazed crops and pastures. Research is also being conducted into the use of dairy effluent in drip-irrigated woodlots, amenity horticulture and so-called fodder factories.

Advantages of subsurface drip irrigation:

- water is applied to plant root zone
- reduced water losses from evaporation
- minimal contact of humans and stock with effluent
- low percolation losses
- minimal land-forming
- minimal removal of trees.

Disadvantages of subsurface drip irrigation:

- increased risk of clogging; filtration is essential

- pumping costs
- installation cost
- energy use
- high maintenance
- potential for damage by stock
- controls depth of root zone.

Localised irrigation

Localised irrigation systems apply effluent directly to plants. The effluent is not distributed over soil where there are no plant roots. Examples of localised irrigation systems include drip irrigation, bubblers, micro-sprinklers and porous pipes. Perforated pipe or micro-sprinklers on the soil surface drip or spray water at the base of individual plants to adequately wet the root zone. Application rates can be monitored to meet evapotranspiration needs and so minimise percolation losses.

The delivery of dairy effluent via localised irrigation systems is of limited value unless the effluent is highly diluted and filtered. As water is concentrated around a plant root zone, it is important that the salt concentration does not pose a threat to the plants. Dairy effluent may need to be shandied to reduce the likelihood of toxicity to plants and to reduce the clogging potential. Localised irrigation systems have, however, been successfully used with saline effluent and are most suited to row crops and fruit production. An application efficiency of 75% to 85% is commonly achieved by drip and micro-sprinkler irrigation.

Advantages of localised irrigation:

- water applied only to the plant
- less water required
- minimises losses to evaporation
- easy to automate.

Disadvantages of localised irrigation:

- high energy use
- high capital cost
- risk of clogging
- corrosion of components.

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3.8 Effluent distribution and irrigation systems

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3.9 Hydraulic application rate and scheduling

Hydraulic application rate can be interpreted as:

- the instantaneous application rate which allows infiltration without runoff
- the amount of water per application to fill the soil profile, or
- the seasonal or annual application needed to meet plant water requirements.

In the case of dairy effluent, nutrient loading usually governs the area of land required for spreading and the water demand by crops is usually satisfied by rainfall or irrigation. However, all of the above points are critical: the first two to avoid the loss of nutrients in runoff, the latter for its limitation on yield and nutrient uptake.

Typical values of hydraulic conductivity

The instantaneous application rate under irrigation must not exceed the soil's hydraulic conductivity (infiltration rate) or runoff will occur. Typical values for hydraulic conductivity are given by Hazelton and Murphy (2007) and are reproduced in Table 1 (these values are estimates only and should be used with caution). The acceptable range for effluent is generally 5 to 50 $mm{\cdot}h^{-1}$.

Table 1. Typical hydraulic conductivities of various soil types (Hazelton and Murphy 2007).

Texture	Structure	Infiltration	Permeability ($mm{\cdot}h^{-1}$)
Sand	apedal	very rapid	>120 (can be >250)
Sandy loam	weakly pedal apedal	very rapid rapid	>120 60–120
Loam	peds evident weakly pedal apedal	rapid moderately rapid moderately rapid	60–120 20–60 20–60
Clay loam	peds evident weakly pedal apedal	moderately rapid moderate slow	20–70 5–20 2.5–5
Light clay	highly pedal peds evident weakly pedal	moderate slow very slow	5–50[a] 2.5–10 <2.5
Medium to heavy clay	highly pedal peds evident weakly pedal	low to moderate very slow very slow	2.5–50[a] <5 <2.5
Clay		moderate very slow extremely slow	8 <2.5 <1

a: Strongly structured polyhedral subsoils, e.g. Krasnozem or Dermosol.

The application rate should be governed by the steady-state hydraulic conductivity of the soil, which can be influenced by groundcover and previous soil management. Over time, the hydraulic conductivity of soil under effluent and solids application can change as a result of pore clogging (Magesan *et al.* 1999). Increasing soil sodicity associated with the application of salt can also reduce the hydraulic conductivity of clay soils.

Soil water-holding capacity

As most plants extract water directly from the soil, the physical characteristics of the soil influence the quantity and availability of water to plants. Soils capture and hold water

within the air spaces and around the soil particles; the strength with which soils retain this water depends on soil structure and texture. This strength is expressed as pressure or suction, with higher values indicating less readily removed water. Water availability in soils is commonly expressed as mm water per metre of soil depth; this relates directly to volumetric soil moisture content.

Typical relationships between soil water tension (the holding strength) and available soil water are presented in Table 2. This information is commonly used to assist scheduling of effluent applications, as soil moisture meters frequently rely on parameters related to it. In conjunction with rooting depth for the crop or pasture to which effluent is applied, it also governs the maximum depth of water that can be applied without having water (and potentially any soluble nutrients) move beyond the root zone.

Table 2. Relation between soil water tension (in kPa) and available soil water (in mm·m^{-1} soil depth) (Doorenbos and Pruitt 1984)**.**

Soil water tension (kPa):	20.3	50.7	253	1520
Soil texture	Available soil water (mm·m^{-1})			
Heavy clay	180	150	80	0
Silty clay	190	170	100	0
Loam	200	150	70	0
Silty loam	250	190	50	0
Silty clay loam	160	120	70	0
Fine-textured soils	200	150	70	0
Sandy clay loam	140	110	60	0
Sandy loam	130	80	30	0
Loamy fine sand	140	110	50	0
Medium-textured soils	140	100	50	0
Medium fine sand	60	30	20	0
Coarse-textured soils	60	30	20	0

Seasonal water demand

Crop and pasture yield, and consequently nutrient uptake, depend on moisture availability. For dairy farms without irrigation, the size of the reuse area must therefore take into account the variation in yield with dry, average and wet years (10th percentile, mean and 90th percentile rainfall years).

Chapter 2.6 ‘Effluent storage requirement’ provides a model for assessing the hydraulic balance and determining seasonal or annual water demand. The following section provides further information on typical crop water use or evapotranspiration.

Crop evapotranspiration

Seasonal water use by crops and pastures (crop evapotranspiration, or ET_c) commonly falls within the ranges shown in Table 3 (see chapter 2.6 ‘Effluent storage requirement’ for calculating ET_c).

Table 3. Range of seasonal ET_c (Doorenbos and Pruitt 1984).

Crop	Seasonal ET_c (mm)	Crop	Seasonal ET_c (mm)
Deciduous trees	700–1050	Tomatoes	300–600
Maize	300–600	Vegetables	250–500
Onions	350–800	Lucerne	700–1100
Oranges	600–950	Perennial pasture	600–900
Potatoes	350–625		

3.9 Hydraulic application rate and scheduling

Water requirements may be met by both irrigation and rainfall, although the latter is unpredictable. The selection of crops and pastures to be grown across the reuse area must take into account water availability, or their capacity to use the nutrients applied will be curtailed.

Crop coefficients

In assessing water use, it is important to take into account crop growth stage and cultural practices such as grazing or cutting. FAO (1988) provides further information on calculating ET_c under non-standard conditions.

Table 4 documents crop coefficients (k_c) for a range of crops. The hydraulic balance model (Table 2, chapter 2.4 'Effluent storage requirement') can be used to indicate the relative water use by a range of crops where the district evaporative rate is known. If reference crop data for the district is available, it will generally be more reliable than modelled data.

Table 4. Indicative crop coefficients (k_c) for water budget modelling (Doorenbos and Pruitt 1984).

Percentage of crop-growing season:	0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
Lucerne	0.55	0.60	0.70	0.80	0.90	0.95	0.95	0.95	0.90	0.80	0.65
Beans	0.20	0.30	0.40	0.65	0.85	0.90	0.90	0.80	0.60	0.35	0.20
Citrus and avocados	0.50	0.45	0.45	0.45	0.45	0.45	0.50	0.55	0.60	0.55	0.50
Maize	0.20	0.30	0.50	0.65	0.80	0.90	0.90	0.85	0.75	0.60	0.50
Cotton	0.10	0.20	0.40	0.55	0.75	0.90	0.90	0.85	0.75	0.55	0.35
Fruit, deciduous	0.20	0.30	0.50	0.60	0.70	0.75	0.70	0.60	0.50	0.40	0.20
Fruit with cover	Averages about 1.00 for periods of rapid growth of cover crop										
Grain sorghum	0.20	0.35	0.55	0.75	0.85	0.90	0.85	0.70	0.60	0.35	0.15
Grain, spring	0.15	0.20	0.25	0.30	0.40	0.55	0.75	0.85	0.90	0.90	0.30
Grain, winter	0.15	0.25	0.35	0.40	0.50	0.60	0.70	0.80	0.90	0.90	0.30
Grapes	0.15	0.15	0.20	0.35	0.45	0.55	0.55	0.45	0.35	0.25	0.20
Shaftal clover	Averages about 0.95 for maximum growth										
Walnuts	0.30	0.35	0.55	0.70	0.75	0.75	0.75	0.65	0.55	0.30	0.15
Pecan nuts	0.35	0.45	0.55	0.75	0.75	0.65	0.50	0.45	0.40	0.35	0.30
Peanuts	0.15	0.25	0.35	0.45	0.55	0.60	0.65	0.65	0.60	0.45	0.30
Potatoes	0.20	0.35	0.45	0.65	0.80	0.90	0.95	0.95	0.95	0.90	0.90
Rice	0.80	0.95	1.05	1.15	1.20	1.30	1.30	1.20	1.10	1.90	1.50
Sugar beets	0.25	0.45	0.60	0.70	0.80	0.85	0.90	0.90	0.90	0.90	0.90
Sugar cane	Varies from 0.55 to 1.00 depending upon rate and stage of growth										
Vegetables, deep rooted	0.20	0.20	0.25	0.35	0.50	0.65	0.70	0.60	0.45	0.35	0.20
Vegetables, shallow rooted	0.10	0.20	0.40	0.50	0.60	0.60	0.60	0.55	0.45	0.35	0.30

Scheduling irrigation frequency

Because water use will vary daily with weather conditions, the frequency with which irrigation water will be applied will vary substantially. Daily water use by some crops can exceed 10 mm on some summer days so much greater reliance is now placed on short-term weather forecasts and soil moisture monitoring

The following equation relates irrigation interval to ET_c and the available soil water from 'Soil water holding capacity' above:

$$\text{Irrigation interval (day)} = \text{available soil water in root zone (mm)} / ET_c \text{ (mm day}^{-1}\text{)}$$

3.9 Hydraulic application rate and scheduling

It is unlikely, and undesirable, that effluent be applied at every irrigation, as effluent storages are generally drawn down over the irrigation months. Effluent applications should be planned for periods of maximum crop growth and to allow a minimum exclusion period of 2 to 5 weeks before any grazing occurs (see chapter 3.11 'Microbial risks').

When applying effluent, whether it be as part of an irrigation event or as the only source of applied water, check the short-term weather forecast to avoid applications before wet weather.

References

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3.10 Land application of manure and pond sludge

Manure can be readily used as fertiliser. As a result of increasing resource scarcity and fuel costs, the price of chemical fertilisers is increasing, in turn increasing the value of the nutrients in manure. However, the value of manure is not just limited the macro and micro-nutrients, but extends to the organic matter, which has beneficial effects on soil health and structure.

The level of dairy farm experience with successful manure and sludge reuse, as well as processing of manure and sludge, are growing. However, environmental policies developed to minimise risk tend to limit the off-site use of solids. With obvious opportunities for using manure to support pasture and crop production, land application is currently the favoured option for reuse.

Accumulation and collection of solids

The milking shed is not the only part of a farm where manure accumulates: feedpads, calving pads and loafing pads accumulate manure too. Whenever animals congregate and generate manure at levels in excess of the capacity of a site, the manure must be harvested and reused. Surfaces that drain to effluent storages or sumps will export manure too. This accumulated manure, solids or sludge will need to be removed periodically. Of key concern in harvesting solids for reuse is the presence of extraneous material such as rocks, clay and sand, and care is required to limit contamination. More information is provided in chapter 2.8 ‘Desludging and pond closure’.

Storage of solids

The amount of manure or sludge yielded and how quickly it is harvested obviously exert major control on the prospects for solids storage. However, typically, solids are generated year-round but are applied only sporadically, and so need to be stored. Storage has the potential for generating contaminated runoff, odours and groundwater contamination. It is preferable to have a dedicated area or structure for the storage of solids where they will have minimal impacts. It is critical to manage any contaminated runoff emanating from storage and handling. Runoff can emanate from the product, from rainfall ingress or from surfaces covered with spilt solids. All contaminated runoff around the solids storage and works areas should be contained. Runoff is typically directed to the dairy shed effluent system. The storage area needs to provide room for the movement of vehicles bringing and taking materials. Solids storage provides a potential breeding or feeding ground for flies, rodents, birds and microbes, so these must be excluded or controlled.

Because the land application of moist material can present difficulties, solids can be dried in heaps, windrows or broad areas. Research is currently under way to examine techniques for drying to increase nutrient density. A dewatered product obviously weighs less and takes up less room for transport and application.

Transporting solids

Transporting solids can lead to complaints, particularly in towns and cities. Loads must be covered to minimise the risks of spillage and complaints.

Land application

Nutrient loadings

Typical phosphorus levels in manure range from 0.3% to 2.0% of total solids, compared with 9% in single superphosphate and 19% in triple superphosphate. The potassium content ranges from 0.3% to 3.0% of total solids, compared with 52% in potassium chloride. Comparisons with nitrogenous fertilisers are difficult to make, but assuming a total Kjeldahl nitrogen concentration of 2.0% to 4.0% of total solids, this is less than one-tenth of the nutrient content of urea (which contains 47% N) on a dry-matter basis. In addition, the moisture content of manure and sludge is often significant, so these materials have a relatively low nutrient density and require high application rates to meet nutrient requirements. When the variable nature of organic nutrient availability is taken into account, there is often a tendency to apply solids too intensively, sometimes resulting in a smothering of the pasture or crop or an inability to incorporate the solids into the soil.

Application rates

Application rates are usually determined by phosphorus or nitrogen loadings (see chapter 3.1 ‘Nutrient budgeting’), although the presence of heavy metals and pathogens can restrict reuse options (see chapters 3.5 ’Trace elements’ and 3.11 ’Microbial risks’). The concentrations of nutrients in manure and their availability are highly variable. Although pond sludge is generally stabilised to non-volatile, non-odorous material after extended storage, manure can still be dominated by putrescible organics, particularly when it has recently been harvested: age and moisture content can markedly influence characteristics. A lack of reliable data on manure and sludge characteristics dictates the need to adopt a very conservative approach to land application.

Spreading

The limitations and availability of spreading equipment pose major impediments to solids reuse. Only larger farms have manure spreaders, whereas typical farms rely on contract spreaders or their own gypsum or super spreaders, which block if the manure is clumped or too wet. By far the most common technique for land application is the use of a tipping trailer or dumper, which applies manure in dollops across a paddock. The manure is then incorporated during cultivation. Chain-activated spreaders are more effective for conveying manure and sludge with a dry matter content over 30%. Tanker transport with soil injection is another option, relying on the use of material with a dry matter content of 15% or less. Liquid application to the land surface from a tanker followed by spreading is a further option. The major impediment to this practice is the need to apply a large amount to a small area during a short time frame. Table 1 shows the association between solids content and handling method. Manure and sludge with a solids content of between 20% and 30% are defined as semi-solid, warranting the use of scrapers, loaders and muck spreaders.

Table 1. Association between solids content of manure and sludge and conveyance method.

Type	Solids content (%)	Handling methods
Liquid	1–10	Gravity flow, pump, tanker
Semi-solid (‘wet’ solids)	8–30	Conveyor, auger, truck transport (watertight box), solid-waste hopper
Solid (‘dry’ solids)	25–80	Conveyor, bucket, truck transport (box)

Other options for spreading include:

- application of dewatered material and incorporation by cultivation

- application of liquid to the land surface by tanker
- application of liquid by subsurface injection
- spreading with a vacuum tanker and spray line
- high-capacity solids-handling pump to high-flow effluent sprinklers
- excavator to drying bed and truck application of dry material
- excavator to dump truck with land application by tipping.

Organic matter loading and degradation

Whenever effluent enters a farm channel, dam or pond, aquatic microorganisms build up and consume the organic matter and the dissolved oxygen. Similarly, land application of manure with elevated COD or BOD will reduce the oxygen level in the soil. Low levels of soil oxygen inhibit plant growth and may produce unpleasant odours (associated with anaerobic conditions). Maintenance of aerobic conditions in the plant root zone under wastewater irrigation is best achieved through sound scheduling practice.

Guidelines for the disposal of effluent to land include a limit on maximum BOD loading. The NSW EPA's BOD limit is 300 $kg \cdot ha^{-1} \cdot week^{-1}$. The ability of a soil–plant system to assimilate biodegradable organic matter depends on the maintenance of aerobic conditions for rapid microbial degradation. Plant roots also must have oxygen; the requirement varies widely among species. For oxygen levels to be maintained, the BOD must be balanced by oxygen diffused from the atmosphere at a rate that depends on the concentration gradient and the oxygen diffusion coefficient, the latter of which depends on soil characteristics and moisture content.

With good site management, the soil–plant system has a considerable capacity for assimilating organic waste. The organic matter in manure adds to the soil organic matter levels directly and through enhancing plant production and the building of microbial biomass. This organic matter is valuable for maintenance of a healthy soil and protecting the soil from erosion. It also improves soil cation exchange capacity, which reduces heavy metal availability and buffers sodicity.

Accumulation

Heavy metals, which are relatively immobile and do not degrade, can be permitted to accumulate in soils until a critical concentration is reached; they are not commonly present at high concentrations in dairy manure and sludge (see chapter 3.5 'Trace elements'). The critical level is usually associated with phytotoxicity or constraints determined by material entering the food chain. Phosphorus applied as a component of manure or sludge can also accumulate but is rarely implicated in plant toxicity. Soil pH is a major controller of the bioavailability of heavy metals, most of which are available to plants at neutral to low pH levels. Increasing pH allows for the accumulation of heavy metals.

Migration of manure components

Waste constituents can migrate with water movement. The capacity of a soil to assimilate them is based on using enough land to ensure that the concentration of the constituent in the recipient groundwater or surface water conforms with established water quality standards. Nitrate is prone to migrate from manure-treated land that is not actively growing a crop or pasture. However, salts must migrate with runoff or leachate to maintain optimum conditions for growth on the application sites.

Crop use

Land use options are many and varied. Table 2 shows typical rates of macro- and micronutrient uptake by a range of crops. The land-limiting constituent dictates the area of land required for a balance (see chapter 3.1 'Nutrient budgeting'), and top-up fertilisers are then required to meet deficiencies in other nutrients. Tabulated data on crop nutrient uptake rates are indicative only, being highly influenced by management and yield, and additional amounts are often applied to cater for lack of homogeneity, lack of uniformity of application, gaseous and leaching losses, and limitations on nutrient availability in soils. Only nutrients in the harvested portion of the plant are listed in the table.

Table 2. Nutrient requirements for broadacre crops in kg·ha^{-1}(FAO 2000).

Crop	Yield (kg·ha^{-1})	Nitrogen (N)	Phosphorus (P)	Potassium (K)	Ca	Mg	S
Rice (paddy)	3000	50	11	66			
	6000	100	22	133	19	12	10
Wheat	3000	72	12	54			
	5000	140	26	108	24	14	21
Maize	3000	72	16	45			5
	6000	120	22	100	24	25	15
Potatoes	20 000	140	17	158	2	4	6
	40 000	175	35	257		23	16
Onions	35 000	120	22	133	–	–	21
Tomatoes	40 000	110	13	125	–	17	54
Cucumber	35 000	60	20	83	–	36	–
Lucerne (hay)	7000	215*	26	108	164	19	19
Soybeans	1000	160*	15	66	–	–	–
	2400	224*	19	81	–	18	–
Beans	2400	155*	22	100	–	–	–
Cotton—seed	1700	73	12	46	6	4	5
—lint	5000	180	27	105		35	30
Tobacco (dry leaf)	1700	90	10	107	48	6	4

–: Data not available.
The table lists plant nutrients contained in the above-ground part, and in the below-ground harvested portion where appropriate, at the indicated yields. Note that these are not the same as fertiliser requirements.
* Legumes can secure most of their nitrogen from the air.

Plant harvest is the most desirable way of exporting the nutrients in manure, but not all of the nutrients are available directly after application, and some can take years. Usually there is an upper limit to the concentration of a constituent within the plant–soil system below which there is no adverse plant response. Once it is applied, no further application should be made until the concentration in the soil has been reduced to about 10% to 20% above the natural level.

References

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3.11 Microbial risks

Dairy wastewater, both raw and treated, contains pathogens that may reinfect the herd or, in some cases, cause disease in humans. Proper management during reuse and the use of exclusion periods before grazing are necessary to prevent infection.

Pathogens of relevance to the dairy industry

Faecal and other wastes (urine, respiratory secretions, sloughed skin etc.) collected by waste management systems contain large numbers of pathogens (disease-causing microorganisms). Such pathogens can cause diseases in animals grazing on the pasture or crops to which manure and effluent have been applied, and to humans via occupational exposure, or via exposure to contaminated water, food, air or soil. The term ‘zoonoses’ is used to describe those microorganisms of animal origin that can cause diseases in humans.

The list of pathogens found in dairy shed wastewater is long; detailed descriptions are given by Pell (1997) and Sobsey *et al.* (2006). In summary, pathogens can be grouped into viruses, bacteria, fungi and parasites (protozoa and helminths).

Pell (1997) nominated the pathogens of most concern to the US dairy industry as *Salmonella* spp., *Escherichia coli*, *Listeria monocytogenes*, *Mycobacterium paratuberculosis*, *Cryptosporidium parvum*, and *Giardia* spp. as a result of confirmed or suspected links to outbreaks of disease in humans. Houlbrooke *et al.* (2004) reported *Campylobacter jejuni* to be the principal bacterial hazard for drinking water and recreational water users in New Zealand. Table 1 summarises the characteristics of these pathogens.

Table 1. Pathogens relevant to the dairy industry (Sobsey *et al.* 2006).

	Pathogen	Disease in dairy cattle	Disease in humans	Transmission
Bacteria	*Salmonella* spp.	May be asymptomatic	Yes	Food, water, and clothing
	Escherichia coli	No	Yes (pathogenic strain O157:H7)	Food and water
	Listeria monocytogenes	May be asymptomatic	Yes	Food, water, and clothing
	Mycobacterium paratuberculosis	Yes (Johne’s disease)	Uncertain (possible link to Crohn’s disease)	Respiratory
	Campylobacter jejuni	No	Yes (gastroenteritis)	Food and water
	Leptospira spp.	Yes	Yes	Urine
Protozoa	*Cryptosporidium parvum*	May be asymptomatic	May be asymptomatic	Ingestion of water
	Giardia spp.	May be asymptomatic	May be asymptomatic	Ingestion of water

Longhurst *et al.* (2000) measured faecal coliform concentrations of 3×10^5 to 1.6×10^6 g^{-1} in untreated dairy shed effluent in New Zealand, and Wang *et al.* (2004) recorded 1.2×10^7 cfu·g^{-1} from fresh dung. Ross and Donnison (2003) found *Campylobacter jejuni* at 10^5 to 10^6 organisms per 100 mL in untreated effluent and 10^3 organisms per 100 mL in a storage pond in New Zealand.

Pell (1997) states that young animals are the most likely animals in a herd to be infected with the range of pathogens listed in Table 1.

Effects of treatment, storage and reuse on pathogen survival

Pathogen numbers can be reduced by two processes: inactivation (e.g. by heat during composting), physical removal or both (e.g. die-off in ponds and sedimentation to pond sludge). Separated solids and sludge must be considered as a source of pathogens during desludging and reuse.

Although there is a body of research into pathogen survival in municipal wastewater treatment, the efficacy of animal wastewater management systems at reducing pathogen viability requires more research. However, Sobsey *et al.* (2006) suggest that:

- *Salmonella* can be detected in liquid manure after 140 days at 10 °C, and *Listeria* after 106 days during winter (durations longer than the hydraulic residence time of some pond systems)
- anaerobic ponds at piggeries may reduce bacterial and viral indicator organisms by 1 to 2 log (90%–99%), but faecal coliform concentrations of ~100 000 cfu per 100 mL remain
- pond efficacy is not consistent and is affected by ambient temperature
- pathogen reduction is consistently improved by the use of multiple ponds in series rather than one large pond of the same volume (minimising short-circuiting)
- pathogen reductions following land application are ‘highly variable and largely unknown; potentially high’.

Generally, pathogen numbers are reduced by sunlight (UV radiation), drying, high temperatures, and high or low pH. Pathogen viability, or more importantly die-off, depends on climate and is therefore difficult to pinpoint. In addition, under given conditions, different pathogens have varying levels of resistance to environmental stresses. Guan and Holley (2003) suggest that the time required for pathogen numbers to return to background levels under dark incubation conditions ranged from 3 days (*Campylobacter*) to 56 days (*E. coli*) under warm conditions (20–37 °C), and longer under cold conditions.

Some factors are contradictory. Although rainfall favours bacterial survival, it may also physically remove (wash) the residues of effluent from the vegetation before any subsequent grazing and reduce the likelihood of ingestion and infection. Vegetation density and height also determine the microclimate into which the pathogens are placed upon reuse.

Composting provides a process-oriented approach to pathogen control. Excepting thermophilic microorganisms, pathogens cannot withstand temperatures above 55 °C for an extended period of time. Composting is therefore particularly effective in reducing pathogen viability in manure solids where all parts of the compost pile are heated. AS 4454 (Standards Australia 2003) recommends achieving 55 °C for a minimum of 3 consecutive days. However, as the outside of the pile remains cooler than the inside, US-based composting standards require a minimum of 15 days in a windrow turned five times for pathogen control.

Given the range of manure storage and treatment practices available to farmers, it is advisable to assume that a significant number of organisms remain viable at reuse and are applied to crop or pasture, and that a withholding or exclusion period is needed to prevent repeated herd infection (see below).

Management practices to reduce risks to stock

Drying, solar radiation and competition from soil bacteria following the application of animal wastes to soil and vegetation can all greatly reduce pathogen populations. However, without proper management, there is significant potential for pathogens to

cause disease in grazing stock. Although cattle avoid grazing immediately around dung pats, they have little choice about where effluent has been uniformly applied.

Recommendations contained in existing state-based guidelines generally include the following strategies:

- Apply wastewater thinly and uniformly to recently grazed pasture so that pathogens can be exposed to maximum sunlight and desiccation.
- Exclude cattle from the reuse area for 2 to 5 weeks.
- Do not apply effluent to paddocks on which stock <12 months of age will graze.

Exclusion periods of 2 to 3 weeks are supported by New Zealand research. Longhurst *et al.* (2000) applied 14 mm of untreated wastewater (equivalent to 25 kg N ha^{-1}) to plots at intervals of 25, 20, 15, 10 and 5 days before grazing. Although faecal coliform counts on pasture had decreased to background levels by 10 days, cows offered a 'taste panel' of plots showed a dislike for pasture treated within the previous 10 days. There was no significant difference between the control and treated pastures at 15 days, but Longhurst *et al.* (2000) recommended a minimum exclusion period of 20 days to maximise intake.

It is clear that some pathogens may still be present at higher-than-background levels after the exclusion period depending on environmental conditions after spreading. However, vegetation growth over that period should minimise the ingestion of soil with any remaining pathogens.

Further information specific to the prevention of Johne's disease can be found at http://www.dairyaustralia.com.au/content/view/273/257/. Two of the recommendations are relevant to effluent management:

- **'Management of the calf rearing area should ensure that no effluent from animals of susceptible species come[s] into contact with the calf.'**
- **'Calves up to 12 months should not be reared on pastures that have had adult stock or stock that are known to carry BJD [bovine Johne's disease] on them during the last 12 months.'**

Contractors should ensure that all equipment that has been in contact with effluent and manure is thoroughly cleaned before leaving the farm.

Aerosols

Pathogens may be transported away from the reuse site on aerosols. The likelihood of such bioaerosols causing human infection is difficult to quantify owing to the variable size of aerosols and weather conditions affecting both the transported distance and pathogen survival. Sobsey et al. (2006) summarises a small number of research papers by stating that insufficient research has been conducted to quantify the risk that effluent reuse poses to nearby residents.

Environmental guidelines for the pork industry (Australian Pork 2006) suggest that 'relatively small separation distances (e.g. 125 m at wind speeds of 0.5 $m{\cdot}s^{-1}$ and 300 m at wind speeds of 2.5 $m{\cdot}s^{-1}$) were needed to minimise any health risks from campylobacter and salmonella in the irrigation aerosols.'

OH&S implications

Given the likelihood that some zoonoses will remain in effluent at the time of reuse, operators and staff must observe appropriate hygiene practices (see chapter 6 'Occupational health and safety').

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4.1 Yards and laneways

Holding yards, tracks and laneways need to be well planned to cater for intensification and for increasing regulation of the environment. A good, well-drained foundation is essential. The planning should involve sourcing information from other farmers, farm planning consultants, dairy system designers and earthmoving contractors. Involving these people in the decision-making process can be advantageous, as strangers to a site can come up with innovative solutions.

Surface conditions and the resilience of the surface under stock traffic are usually overemphasised in extension notes. This overemphasis derives from the role of surface conditions in lameness, hoof damage and all-weather access during wet weather. Better practice favours an appreciation of farm drainage and optimising the use of earth available on farm before using pavement or surface material.

To assist farm planning, an aerial photograph of the farm, a topographic plan with a contour interval of 1 m and a soil map are invaluable; if available, a geophysical survey plan is also useful (this is usually referred to as an EM survey plan). Geographical information system (GIS) data can be used to calculate the area of subcatchments on a farm to assist drainage design. Farm planning is a useful group exercise, particularly if neighbours are involved: the involvement of neighbours offers better prospects for integration of plans to control erosion and divert contaminated runoff. If waterways traverse the property and are crossed by roads, the size of culverts used by local road authorities can serve as a gauge for farm works. If possible, runoff should be delivered under or around holding yards, tracks and laneways via diversion banks, drains and culverts. The less water on the facility or draining off it, the easier will be surface management.

Planning approval may be required for farm earthworks, particularly if they are sited on a flood plain. In Victoria, excavations to increase farm water storage capacity need the approval of the regional water authority. Earthworks can lead to substantial landscape modification.

Laneway alignment

Where a new track or laneway is proposed or yards need reconstruction, the following points should be considered.

Terrain

The information obtained from an overview of the terrain is invaluable since the direction of the natural drainage can be studied. The best location for a track or laneway is obviously one which is not too steep, not liable to flood and permits all-weather access. In the alignment of all tracks and laneways, certain definite localities, or fixed points, on the alignment need to be set (e.g. a gap in a line of hills, the best point on a watercourse for a bridge, the best point on a stream for a ford or culvert, soaks or spring lines, soft ground, trees to avoid, shaded areas). The fixed points are determined first, and the track or laneway is then designed to run as directly as possible between one fixed point and the next. However, the laneway will not necessarily run in a straight line between these points, as a laneway should follow ridge alignments or catchment divides. This allows the natural drainage of the area to fall away from the laneway, reducing the cost of bridges and culverts and resulting in a more stable construction. Where hills or spurs obstruct or project into the line of the laneway, it is usually better to go around them rather cut through them. It is useful to allow a deviation equal to 20 units in length for every unit in height of the hill avoided. For example, if a hill is 10 m high, up to 200 m might be added to the length of the

laneway to avoid it. Otherwise, the straightest and shortest route is the cheapest and best.

Natural stock movement

When investigating the route of a new track or laneway, examine stock movement patterns in detail, as it is easier to move stock along an alignment they have naturally selected, providing this is compatible with other requirements and with overall farm planning. This can involve the avoidance of hills and natural obstructions.

Drainage

Drainage is discussed in detail below. However, in siting, it is important to note all watercourse and drainage line crossings and the extent of any areas liable to flooding or inundation.

Slope

Set and adhere to the maximum and minimum slopes for tracks and laneways. The maximum slope should be 1 in 10 (10%), but its length must be short; for a long section, the maximum slope should be 1 in 20 (5%). Steeper gradients of 20% or even 33% can be encountered in mountainous terrain but should be avoided if at all possible.

Switchbacks

Switchbacks might be necessary in steep country, but wherever possible stock movement should avoid these. Usually dairy cows like to go in the same direction and to follow a defined movement pattern, which can be disturbed if animals appear to be going against the flow.

Foundation

Aim to source the material to serve as the surface and subsurface components of the laneway on site.

Shading

Observe the location of trees and sheds and the way these influence shading, as shading and tree roots can cause deterioration of a laneway.

Width

Take every opportunity to observe the proposed line of the laneway from adjacent vantage points, making sure that adequate width is allowed. Kilndworth *et al.* (2003) provide recommendations on the configuration for a range of herd sizes.

Drainage

An integrated drainage system for holding yards, farm tracks and laneways is essential; appropriate drainage may render a bad track good and a poor surface viable. It is necessary not only to divert the runoff from the surface of a track or yard, but also to block runoff from adjacent surfaces. If the track is well sited, the natural drainage of the land may be sufficient, as for example a track on a ridge. In any case, interfere with the natural drainage as little as possible. As the resistance of a track surface depends on the moisture content of the soil (the track must not be too wet or too dry), the degree to which drainage should be carried out has to be assessed. Surface and subsurface

drainage techniques are explained below. The main concerns are cost and the fate of the runoff.

Surface drainage

A topographical map with contour plans can be used to plot a trial alignment. In rolling country, a design gradient range of 2% to 5% is usually appropriate. Avoid a flat laneway section unless it has an adequate camber or cross-fall to parallel drains; the desirable minimum design gradient of a laneway is 0.02%. In flat country, excavation is often required to build up the laneway so as to aid drainage, preferably with a cambered surface. Borrow areas should be located close to the proposed route of the laneway and capture runoff from it. Standard road design practice can be used for aligning laneways on large farm plans, particularly if future subdivision is mooted.

Surface water passes from the track to shallow gutters, and thence through dished drains to table drains. For those parts of the landscape where the surface falls away from the track, table drains can be omitted, and the gutter can drain into the paddock. Sandy soils do not need much surface drainage. Where table drains are necessary, and that is usually apparent, they must not be built at the edge of the track: allow at least 3 m (or, better, 5 m) between the track and the drain. Table drains (ditches) should be wide and shallow, and preferably vegetated and capable of being mown. The soil excavated from them should be spread well back so that it does not wash back into the drain. Table drains should spill into watercourses or drainage lines or into low places which fall away from the track or laneway. Sediment traps may be required; pollution control agencies encourage these in association with wetlands, or at very least diversions to paddocks rather than direct discharge to a watercourse.

The gradient of a drain is important: if it is too steep, the water will erode a drain into a deep chasm; if it is too flat, silting will occur. Guidelines on drainage design can be found in Underwood (1995). Design criteria are now under review in response to climate change modelling, which indicates that average annual rainfall in southern Australia will decline but higher-intensity fall will increase. If the gradient is too steep, erosion control barriers (or steps) should be built across the drain at frequent intervals to retard the flow and allow sediment to be deposited. Many tracks and laneways have been destroyed because the side drains that were meant to protect them were badly sited, sized and levelled.

Subsurface drainage

Subsurface drainage takes the form of either pipe-less drains (mole drains) or, more commonly, slotted HDPE or uPVC pipes installed below the natural surface. The size and depth are based on the desired depth for water table control, but the spacing is largely dependent on soil profile characteristics and rates of extraction. The pipes discharge by gravity to an outfall or a pumped sump. From here, leachate can be applied to land or discharged to a receiving waterway, as long as nutrient levels are not excessive. The recommended minimum diameter for an HDPE subsurface drainage pipe is 80 mm.

Fate of leachate

Runoff, especially from stock bridges, should not be allowed to discharge directly to a waterway or to a permeable stratum. It is preferable to mount bridges on abutments above the level of the bank and provide a slope to direct the drainage back to the abutments. This runoff should be channelled to sediment traps or paddocks rather than the waterway. Transverse drains are commonly used to direct runoff away from waterways and to direct laneway runoff from bridges.

Where a stock underpass is built, it is essential to locate that a transverse drain at the top of the exit and approach ramps to divert catchment and laneway runoff from it. A

transverse drain consists of either a grated gutter or, preferably, a trafficable bund. Underpass drainage is a specialist area, and as underpasses form a retaining wall, they are commonly designed to weep to provide pressure relief. A dewatering sump is sometime installed to collect subway leachate and direct it back to the natural surface for safe disposal.

Construction

Site preparation

Where a laneway or a holding yard is to be established, clear the site of all trees, shrubs and stumps and pull out all tree roots to a depth of at least 300 mm below the natural surface, and remove them from the site: no vegetation should be incorporated into the earth to be used for construction. Because of its high organic matter content, topsoil is unsuitable for compaction, so all topsoil must be stripped from the surface to a depth of at least 150 mm and removed from the site.

Prepare the foundation to form a satisfactory surface for the placement of layers of selected material.

- Ensure that the surface is well-drained, with a slope exceeding 0.02%.
- Place and compact suitable material into any holes or depressions resulting from the removal of tree stumps and roots.
- Scarify or rip to a depth of at least 150 mm. If the exposed foundation material does not comply with specifications, further excavation will be required.
- Water to suppress dust.
- Compact to increase the density of the foundation material.
- Walk the area to ensure that all foreign material has been removed and to locate any potential areas of concern for subsequent remedial treatment.
- Lay crushed rock, rather than sand, to a depth of 200 mm.
- Ensure that no sharp rock sits proud of the surface.

Constructing the pad

Following preparation of the foundation, the pad (i.e. the compacted fill forming the desired slope) can be constructed. All fill placed on the pad must be within ±2% of the optimum moisture content (specified by laboratory testing or experience) and placed in progressive horizontal layers with a uniform thickness of not more than 200 mm before compaction. Wrigley (1996) provides more information on pad construction. Pads for yards and laneways do not have to provide an impervious surface; their main objective is to provide stable support for concrete and traffic.

Optimum moisture content

The moisture content of all material placed in the pad must be within ±2% of the optimum moisture content required to produce the maximum dry density when compacted in accordance with AS 1289 (Standards Australia 2000).

As a guide, the required moisture content for clay is as wet as can be rolled without clogging a sheep’s-foot roller. The moisture content of clay can assessed by rolling a sample of it between the hands. If it can be rolled to spaghetti thickness without breaking, it should be satisfactory, particularly if it starts to crumble at the ends.

If water has to be added to achieve the required moisture content, add it to the borrow area to allow even distribution throughout the material *before* excavation. To achieve

effective water distribution, rip the surface of the material in the borrow area before watering. Water can be added following placement on the pad, but only when it is not possible to add all the necessary water in the borrow area. Never place dry material on the pad. Generally, if dust is generated during placement of fill, the fill is too dry.

Compaction

Compact each layer of material to produce either a field dry density of at least 95% of the standard maximum laboratory dry density determined in accordance with AS 1289; or a Hilf-density ratio of at least 95% when tested in accordance with AS 1289.

This degree of compaction may generally be achieved in clay by rolling each layer at least eight times with a sheep's-foot roller. As a guide, compaction of clay will generally be sufficient when there is a clearance of 100 mm between the drum of the roller and the compacted material.

Maintenance

Periodically scrape packed manure from the surface of yards and laneways for storage and reuse. But leave a residue rather than expose the surface of crushed rock or clay.

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4.2 Feedpads, calving pads and loafing pads

A **feedpad** permits the supplementary feeding of cows, providing water, space, feed, and effluent and manure removal. Cattle are held or mechanically fed for the purpose of milk production or animal husbandry.

Although there is significant research on feedlots, research on dairy feedpads in Australia is lacking. A **feedlot** is a structure where cattle receive an entire ration for maintenance, weight gain and milk production; in contrast, a feedpad is dedicated to supplementary feeding only.

Feedpads are used to feed hay, silage, mixed rations or concentrates with minimal loss to stock in times of pasture shortage. They also reduce the adverse impact of stock on pastures and soil during wet conditions or pasture renovation. A **loafing pad** can be a component of a feedpad, offering a retirement area for stock to ruminate. Both facilities are supplied with water and sometimes shade. **Calving pads** can be standalone structures or form part of a feedpad complex. In a calving pad, a well drained and sometimes bedded area is provided to enhance the health of calves and their mothers during and after birth.

Both shade and shelter are required if dairy cows are to be exposed to the elements for a long period. Good drainage and harvesting of manure, spilt feed and contaminated bedding are also critical components of pad management. Farmers who use a feedpad aim to maintain a herd and milk flow, increase quality or quantity of milk, and avoid deterioration of tracks or pastures. Other factors are the need to mitigate heat stress and to sustain a herd which cannot be supported through an existing feed base.

Pads need to be planned, designed and built to be economically viable and environmentally sustainable. They also need to comply with animal welfare regulations and accepted community standards for environmental performance. Provision needs to be made for all-weather feed supply, drainage, waste removal from the pad and the remote storage of effluent and manure. Structures should be formed for ease of access and maintenance and to facilitate harvest of manure and contaminated runoff. They must also provide a suitable surface and, if necessary, shelter to prevent pugging in wet weather. Provision of an adequate slope for effluent drainage, ease of animal traffic and movement of mechanical equipment is essential. Diversion of uncontaminated stormwater to drains should be made possible when the pad is clean and not in use.

The planning, design and operation of feedpads, loafing pads and calving pads incorporate a range of objectives:

- safe and easy access for animals, vehicles and farm workers
- adequate room and headspace to provide acceptable conditions for animal production
- the storage of and access to a range of feeds and the use of specialist feeding-out equipment
- availability of bedding material and equipment to renew bedding
- drainage to allow all-weather access and to divert catchment runoff
- collection of contaminated runoff
- harvesting of manure, soiled bedding and spilt feed
- waste storage on impervious surfaces and in well-bunded structures
- use of harvested manure and effluent on crops and pastures
- protection of surface and groundwater resources through prevention of leachate movement and control of runoff

- maintenance of community amenity through odour, noise and insect control.

Principles of waste management

Feedpads can be used on a permanent, regular or occasional basis. They are frequently used during wetter, colder months and just before or after milking, but are more commonly used in response to drought and pasture shortage. Shaded feedpads and loafing areas are also used to relieve heat stress in summer.

Accumulated manure is either cleaned off the pad surface by scraper or front-end loader or is washed off. The design criteria for cleaning a feedpad by floodwashing are similar to those used for cleaning a yard. Runoff contaminated with manure is usually directed to storage before land application. Harvested manure and solids are usually dried for land application.

Calving pads and loafing pads generally incorporate a well bedded and padded surface with well defined surface or subsurface drains. These pads need to be cleaned regularly by front-end loader to remove soiled bedding and manure. Sometimes disinfectant is applied before the supply of new bedding.

The amount of manure generated on the pad depends on the time the animals spend there out of every 24 h (see chapter 1.2 ‘Characteristics of effluent and manure’). In the absence of research it is assumed that the proportion of manure to be managed is related to the time the animals spend on the pad.

Enclosed or covered pads are usually cleaned by floodwashing and scraping. On an enclosed pad or freestall barn the effluent generated depends on the volume of floodwash, the presence of bedding (to absorb effluent) and the amount of washdown water used. An exposed pad has the added burden of rainfall runoff to contain: The volume generated depends on rainfall; usually the volume of effluent storage is dictated by the need to contain runoff from a 24 hour rain event with a recurrence interval of 1-in-20-years.

Regulations

Rules on the development and expansion of pads are ambiguous and so can lead to disputes. The main question is when a broadacre farm becomes an intensive animal enterprise or a feedlot. Under drought conditions or where inclement weather is common, farmers are encouraged to develop structures and systems for supplementary feeding, but local planning ordinances must be followed.

Regulatory requirements for feedpads vary between states, and it is not possible to cover all state requirements here. Planning conditions usually govern buffer distances to neighbouring houses, waterways, groundwater bores, roads and towns; and structures usually need to conform with building standards. Waste management conditions are commonly dictated by the scale of works and usually set requirements for the storage and land application of effluent and manure.

Most of the concerns that planning authorities and neighbours have with intense dairy production relate to odour, noise, light access, heavy vehicle movement, loss of property value and amenity, the fate of wastes and the use of medications. Some animal welfare groups campaign against intensive animal housing, and occasionally the RSPCA gets involved in the regulation of facilities.

Planning and construction

To be effective, a pad needs to be considered as integral to a farm. Integrated farm planning is essential to deciding the site and size of a pad: the whole farm needs to be considered, not just the pad. Landholders proposing to install pads should look beyond the immediate problem that prompted the pad and consider the likely long-term effects

of their actions, and evaluate associated costs and benefits in detail. They must also consider existing and future neighbours, and development prospects for their own and neighbouring properties.

To minimise the energy and labour requirements of a pad, place it near a milking shed or in clear sight of the shed or farmhouse. Avoid placing it in clear view of neighbours or next to a major road, and site it for minimum obtrusion.

Pads should be formed well above the natural surface level to promote drainage, increase air movement and discourage insects. The design should allow for all-weather access by machinery and cows and for waste removal. The positioning of the pad should be dictated by the location of the milking shed’s access track and the holding yard. Pads should be planned as multipurpose facilities which can be used to benefit the farm all year round.

Avoid siting a pad where topography favours katabatic drift to neighbours located downhill or where rural residential development is encroaching. Chapter 5.1 ‘Odour emissions and buffers’ provides detail on odour propagation and control.

Aspect and dimensions

Intensive animal production prompts community concern for the potential for adverse environmental impacts and animal welfare. These factors must be taken into account in the sizing of feedpads to provide adequate facilities and conditions for maintenance of the health of stock and protection of the environment. Case studies are presented in Davison and Andrews (1997).

Dimensions for components of a feedpad are usually set by experience or personal preference, and should allow for herd expansion. The recommended alignment or aspect of feedpads tends to vary from one part of Australia to another; for example, in Queensland a north–south alignment is favoured, whereas in Victoria an east–west alignment is popular.

Siting and sizing

Space requirements for animals on a feedpad are usually a function of how it the pad to be used and how long the animals are to stay on it. When designing any farm facility, consider the physical dimensions of structures and laneways. Adequate areas must be set aside for stock movement and vehicle access. The approaches to and from a feedpad are subject to intensive stock movement. The minimum desirable width for laneways is 3.7 m to facilitate vehicle movement. As herd numbers increase, recommended laneway dimensions rise; for example, in a 120- to 250-cow dairy, 5.5-m laneways are adequate (Wrigley and Phillips 1993). A compromise is necessary between very wide laneways, which occupy land and reduce control of stock movement, and narrow laneways, which funnel stock and contribute to high pavement loading. In the case of feedpads, large trucks will need to bring in feed without excessive backing, so they need adequate vertical clearance too.

Minimum space requirements

As a general guide, a 600-kg dairy cow requires a minimum area of 9 m^2, preferably 15 m^2 (these provisions must be seen as indicative only, given the immense variety of pads in use). (In contrast, a grazing animal requires around 200 m^2.) An allowance of 10% greater than the estimated need will provide flexibility in feed type and ration (ARMCANZ 1997). Obviously, smaller cows require less space, but it is best to adopt the larger size to allow for future needs. Space requirements depend on the availability of land, environmental conditions, management practices, type of housing and the pad construction materials.

Allowance for cattle spacing also has a significant influence on the amount of manure and its moisture content. The incidence of mastitis and other stock diseases is increased by manure accumulation, so it is wise to be generous with space. Cramped conditions can lead to dust, odour, runoff, mud and fly breeding. Generous space allows udder wash and wipe equipment to be used if required.

In such a restricted area, requirements for skilled managers, up-to-date technologies and energy all increase.

Feed troughs

The spacing requirement for feed and water troughs depends on cattle size, type of enclosure, type of feed and feeding frequency. The minimum trough length for a continuously confined 600-kg cow fed once a day is 0.3 m (ARMCANZ 1997). Guidelines from the USA (Mid West Plan Service 2000) allow a feed bunk space of 0.46 m per cow for mixed rations fed several times per day, or 0.66 to 0.76 m per cow for once-a-day feeding. The allowance depends on the type of feeding system: this could range from a wall with a hot wire to a conventional feed trough or a feeder. If too much room is allowed, wastage can increase; the objective should be to allow the animals to feed and then move back.

Water requirements

Cows must have free access to good drinking water. Table 1 provides general guidelines for water consumption. Feed, geography, age, bodyweight and climate have a major influence on the consumption rate. Under drought conditions where no pasture is available and cows must be kept cool to avoid heat stress, these allowances must be seen as minimal: at least 150 L per cow per day is recommended.

It is essential to allow adequate water for stock on a feedpad; the longer the animals are confined, the greater will be the demand for water. The climate and the moisture content of the ration will dictate consumption, so Table 1 is indicative only, and rates must be tempered by local knowledge and experience. If the water is saline, the stock will drink more and the wastewater will be more salty and more difficult to reuse.

Table 1. Stock water consumption.

Body weight (kg)	Average water consumption ($L{\cdot}day^{-1}$)
50	6–7
70	7–9
90	10–11
120	14–16
150	18–25
190	25–35
350	35–40
540–730 (dry cows)	20–40
540–750 (lactating cows)	45–110

Source: ARMCANZ (1997).

The reticulation system should supply at least 20 $L{\cdot}h^{-1}$ per cow to meet the short-term needs of the herd (Holmes *et al.* 1987). The pipe diameter to meet these requirements needs to be at least 75 mm, and the operating head must be at least 10 m. A tank could be used for short-term supply in the event of a power failure. Water troughs should be well separated from feed troughs. Provision should be made for water flow directly into the drainage system and for drainage control in the event of burst mains or a jammed float valve.

To provide an adequate and constant water supply to stock, a circular loop water line can be installed. Troughs should be located on the high point of a water line to reduce

sediment and to facilitate purging air from the pipeline. In pens, troughs may be positioned and shared between adjacent pens.

Dimensions of pads and sheds

- In a freestall barn or covered feedpad, the height of the canopy has to be accessible to machinery to allow cleaning, feeding and maintenance in the enclosure (with a minimum ridge height of 7 m for access by a front-end loader or backhoe). Standard building frames will aid expansion and planning approval.
- The height of the ridge should allow for the stack effect to remove gases via a ridge vent. The right height of the canopy will promote ventilation while protecting stock.
- The minimum height of eaves should be 3.7 m, although this could be adjusted by the need to provide shade.
- A roof pitch of 1:3 (18°) will allow convective heat dissipation.
- Positioning of the eaves of a shed or canopy to catch winter sunlight will provide warmth for stock, help dry the feedpad surface and reduce the incidence of disease.
- Careful locating and positioning of the shed will create summer shade.
- An east–west aspect is encouraged for Victorian covered feedpads.
- Sheds should be sited near feed stores and effluent storages.
- A pen area of 9 m^2 is desirable to allow the movement of stock to and from troughs (although 1.5 m by 3 m is adequate).
- Laneways, races, entrances and exits should be designed to take advantage of the social behaviour and movement of cows. Use only rounded railing without protrusions in areas of stock congestion. Rounded edges on concrete are essential.
- The shed should be readily seen from the house or milking shed.
- Trees planted to shroud the shed should not completely obscure the view.

Surface of the pad

- Dairy feedpad surfaces should provide sufficient slope for effective drainage. A compacted earthen surface needs a gradient of at least 1:500 (0.2%), but a concrete surface can be drain at a slope of 1:2000 (0.5%) or even shallower if smooth. However, operating experience shows that it is better to aim for slopes in the 2% to 4% range. The earthen pens of beef cattle feedlots drain best at a slope of 3%. The achievement of a suitable slope is based on site-specific conditions, and material choice must be determined by the soil type, cost of construction and amount of shade.
- Surfaces should prevent effluent from reaching subsoil.
- Surfaces should have a thick enough foundation to spread loads without settling.
- Surfaces should minimise stress, disease and injury of animals.
- Surfaces should provide a durable, clean working area.
- Surfaces should be designed to be renovated easily and avoid breaking down when in contact with bedding material.

Concrete feeding aprons have been found to reduce odour propagation and feed wastage and to improve animal health and maintenance. The minimum recommended width is 3 m, with a thickness of 125 mm (which includes a reinforcement cover of 50

mm). For both floodwashing and dry scraping, a 3% slope is favoured. Ideally, a one-way diversion will divert effluent away from the earthen surface of the pen; to achieve this, the concrete edge of the apron should be slightly bunded. Alternatively, the concrete can be sloped back to the trough to form a wide gutter for flooding or removal of rainfall runoff.

Feedlot and feedpad profiles (on bare earthen surfaces) usually contain a compacted interfacial layer of manure and soil. This forms a seal that decreases water infiltration into the soil. Low infiltration restricts the leaching of nitrates, salts and ammonium into the subsoil and protects the groundwater from contamination.

Pad construction materials

The materials forming the pad exert one of the most significant controls on pad performance. Ideally the pad surface should be evenly graded and compacted to form a smooth, impervious surface. Materials used for pad surfaces are many and varied. The most common pad surfaces are:

- earth and stabilised earth
- gravel and coarse sand
- bitumen
- concrete
- rubberised mats.

Loafing pads and calving pads are covered with a softer layer such as:

- rice hulls
- straw
- sawdust
- sand
- almond husks and fruit pips.

Frequently, gravel and sand are laid without grading to finer particles. This cohesionless material is then hard to compact to an appropriate density with minimal permeability. Generally, uniform and poorly graded gravel or sand should not be used for pad construction, as manure harvesting is made difficult and the manure contains gravel and sand.

There are advantages and disadvantages to all types of feedpad surfaces. Availability and cost usually govern the choice. Compacted earth is the cheapest but least resilient material, and concrete is the most expensive but the longest lasting. A design which allows for stages in construction from earth to stabilised earth to concrete is often desirable. But whatever material is used, the foundation must be well prepared.

Geotextiles

In recent years the use of geosynthetics for compacted pads has grown. Geosynthetics are a relatively new concept in engineering materials. One of the largest groups of geosynthetics is the geotextiles. These are thin, flexible, permeable sheets of synthetic material used to stabilise and improve the performance of soil in civil engineering works (Ingold and Miller 1988). Functions can include filtration, drainage, separation, reinforcement and moisture blocking. Filtration restricts the migration of fine soil particles while permitting water movement. Reinforcement stabilises the soil and decreases compaction by stock.

The drainage capacity of geotextiles allows water to be carried along the plane of the material to an outlet, either vertically or horizontally. A one-piece polyester envelope

material can be fitted over perforated drainage pipes like a sleeve. Geotextiles have a relatively low capital cost and are resistant to soil chemicals, moisture and bacteria. They offer a viable approach to pad surface construction at a relatively low cost with a long life span, allowing lateral subsurface drainage. A suitable arrangement for separating soil layers and for stabilising, reinforcing and draining a pad uses a compacted earth foundation 200 mm thick covered with 100 mm of sand, then the geotextile mat, and finally a compacted clay-based surface 300 mm thick.

Cleaning and maintenance of pads

Drainage of pads

Runoff from paved feedpads is affected by slope, rainfall intensity and management practices. Feedpad runoff contains relatively high concentrations of nutrients, salts, chemicals, debris, pathogens and organic matter, and must be collected, treated and stored for reuse. Rainfall must be considered in the design of feedpads, especially exposed pads, as it can generate contaminated runoff. Adequate provision is required for collecting runoff from heavy rain. The tank or pond should be able to accommodate at least a 1-in-10-year, 1-h rainfall.

For good drainage, the feedpad slope should be between 2% and 4%. Slopes outside this range may be acceptable but require a higher standard of construction and operational management. If they are too steep, high-intensity runoff and sediment transport may cause erosion and pollution.

The drainage system of a feedpad should incorporate:

- drains or diversion banks
- a sedimentation basin to remove solids from liquid effluent
- catch drains (minimum slope of 0.5%) to carry storm runoff and effluent.

Ideally, open drains should carry feedpad effluent. Large-diameter pipes are also suitable; the larger the diameter, the less slope is required. Avoid grated pits and pipes <100 mm in diameter as they can easily block and are difficult to clean. Information on sumps, pumps and pipes is provided in chapters 1.5 ‘Sump design’ and 1.6 ‘Pipes’. Pads built on a raised platform will promote natural drainage and gravity conveyance of waste, thus avoiding problems of pump blockage.

Removal of wastes from pads

Effluent, manure and soiled bedding must be removed from a pad to maintain a clean surface and reduce pathogen concentrations. Storage will be required if waste is generated during cold or wet weather or at a time of year when land application cannot take place. Wastes include:

- solid or semi-solid dried and packed manure
- effluent
- waste feed
- waste bedding
- entrained earth, manure, spilled feed and bedding in effluent, which need to be separated by gravity or a grille.

Unless solids are frequently removed from the surface of a feedpad, all wastes will end up in the holding pond.

Floodwashing of pads

Floodwashing is popular for cleaning the paved surface of the feed alley and feeding apron of a feedpad. This practice can aid in the recycling of water and reduce the volume of effluent to be stored in holding ponds, thus reducing the required size of storage. The washdown volume must be applied as a surge flow by a high-flow pump or by gravity release from a tank. If recycled effluent is used for floodwashing, at least two ponds should be used, and only the treated effluent from the second pond should be used for yard washdown. See chapter 1.4 ‘Floodwash systems’.

Checklist

Before building a feedpad, consider the following factors:

- the cost of substitute-feeding of cows on a pad versus the cost of pasture productivity decline due to compaction and disturbance of soils
- likely fluctuations in the price of feed and selling price of milk
- the increased handling and movement of stock on feedpads
- space requirements for access of stock to feed and water
- removal of effluent from the pad; rainfall; and effluent storage
- the environment and contamination of ground and surface waters
- efficient planning, siting, construction and management of the feedpad
- potential risks of odour and adverse community reaction.

A typical system and stages in development are presented in Appendix B.

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4.3 Feed storage and wastage

Some basic rules apply to the handling of feedstuffs and the management of the feeding system to minimise wastage and spillage. Of particular concern is the management of odours, contaminated runoff, unwanted degradable solids and refuse associated with the storage and handling of feedstuffs.

Types of feedstuffs

The reliance on pasture grown on dairy farms continues in many districts, but farmers are increasingly relying on both supplementary fodder grown off-farm and introduced rations. Types of feed are many and varied, and their use depends on cost, availability access and preference. Feeds include:

- straw
- hay
- silage
- grain
- pellets
- food by-products; e.g. brewer's grain, palm kernels, marc (the residue left after grape pressing), citrus pulp, chocolate powder
- time-expired or unsaleable food products, such as stale bread, fruit and vegetables, bakery products and confectionery seconds.

Transporting feedstuffs

Dairy farmers need to consider transport routes for by-products to minimise complaints, particularly in cities and towns. The occasional spill from an overturned truck can cause much angst as well as odour. Loose products must be covered to avoid spillage. In some jurisdictions, curfews limit site access. Sites receiving loads must allow ample room for vehicle movement: at least 5 m around bunkers or silos for safe access.

Feed management systems

The major factors dictating system selection are capital and operating costs, the type and availability of feed, and the ability to cope with a range of feeds at the same facility. Harris (1984) provides relatively unbiased coverage. The variety of storage and feeding systems is immense; systems include:

- trench silos
- clamp silos or bunkers
- vacuum stack ensilage
- round bale ensilage
- towers (e.g. Harvestore brand)
- silage baggers (e.g. Silopress brand).

Sites set aside for these systems need allow room for the movement of vehicles.

All of these feeding systems, with their associated watering systems and stock congregation, have the potential to accumulate manure, thereby increasing the risk of surface water and groundwater contamination. Additionally, there can be significant loss

of feed with all of them, further increasing the risk of emissions. The facility manager will need to regularly collect spills, product residues, waste plastic and runoff for reuse or disposal. This practice will not only reduce the risks of surface water or groundwater contamination and odour emissions, but also keep the feed quality high.

Although much is known about the use of manure and effluent from feedlots and the control of atmospheric emissions, details are lacking for sites where animals congregate in unconfined conditions, and feed losses are rarely quantified. The following practices will aid feed management:

- Calculate how much feed will be transported and stored and how often it will be received. Take note of the moisture content.
- Rotate portable facilities.
- Harrow or harvest accumulated manure.
- Feed out at a site where odour emissions, runoff and groundwater ingress are least likely to affect the environment beyond the farm.
- Ensure that access lanes can be cleaned by a back scraper or front-end loader by providing a well-designed surface.

Information on the fodder requirement for dairy cattle, particularly during drought, is widely available (Hinton (1994), Leaver and Grainger (1989), Long (1992), (Freer 2007)). It covers trail feeding, broadcasting, feeders, dispensers for urea–molasses supplements and feedlots.

Feedstuff spills and losses

During the mixing of rations and feeding out, losses will occur. All storage and feeding-out losses contribute to odour, greenhouse gas emissions and solid wastes. Although these feeds do not carry stock or human pathogens, they can contain nutrients and have a high BOD.

Control of feed losses also reduces the pest burden on a farm and potential disease vectors.

Storing silage

Ensilation is an anaerobic fermentation process used to conserve fodder for an extended period with minimal loss of quality. The reliance on anaerobic conditions contributes to the potential risk of odour propagation and biogas generation. Minimising contamination with earth, old silage and water can reduce the risk of emissions during silage making, storage and withdrawal for feeding out. Rapid resealing, covering exposed bales and reducing leaks will keep the period of exposure to the air to a minimum. Suitable conditions can be created by compacting and storing the raw material in a sealed vessel. Any oxygen not removed by compacting is rapidly removed by bacteria.

Sealing prevents CO_2 from escaping and the re-entry of air during storage. Any contact between the silage and air will result in decayed, inedible and sometimes toxic material. Aerobic degradation increases dry material losses and reduces nutritional value. This material then becomes waste.

The quality of fermentation depends on the types of bacteria present in the sealed vessel. Effective fermentation requires the presence of lactic acid bacteria and the absence of clostridial bacteria. Clostridia are inefficient at converting plant sugars to acids, and produce silage of poor nutritional value. Lactic acid bacteria are very efficient at converting plant sugars to acids, and produce the non-odorous fermentation associated with good silage.

When acidity builds up in the vessel, microbial activity diminishes and the plant material is preserved. The removal of residues, spilt silage, old plastic and runoff can reduce the amount of clostridia in the silage, thereby increasing its value.

Improving the effectiveness of ensilation will result in:

- less odour in general
- less noxious odour
- less unusable product
- less liquid, because clostridia rely on wet conditions, whereas lactic acid bacteria rely on plant sap.

Wastage of silage through spills and exposure has both economic and environmental costs.

Losses associated with silage storage and feeding out

Effluent from silage stores can emanate from the product or rainfall ingress, and rainfall from surfaces containing spilt feed can add to the effluent burden. This effluent can have a very high BOD: concentrations over 10 000 $mg \cdot L^{-1}$ have been registered. Regular cleaning of surfaces and separation of entrained solids will reduce treatment requirements and odour emissions. All contaminated runoff around the feed storage and works areas should be contained and reused.

Leachate is associated with the type of storage system: the higher the stockpile or silo, the greater the pressure and the potential for seepage (Ernst *et al.* 1990). Seepage is associated with loss of quality. Table 1 indicates the losses associated with the content of dry matter (Ernst *et al.* 1990).

Table 1. Dry matter content and effluent.

Dry-matter content (%)	Dry-matter loss in effluent (%)
10	12
33	0

Spoilage is associated with edges, usually through poor sealing and air penetration. Rainfall penetration and soil contamination can result in poor uniformity of product. Rats, mice and other vermin and poor quality control during handling can expose silage to air and thus spoilage.

The amount of feeding-out loss depends on the method used: feeding in troughs or on a concrete pad or mat results in less loss than on the ground, along fence-lines and laneways or by self-feeding from the bunker, owing to less soiling and trampling by livestock. The activity of bacteria, yeasts and moulds is accelerated when silage is exposed to air; these microorganisms degrade the silage, producing unpalatable and potentially toxic feed of low nutritional value. Exposure of silage during feeding out is a major cause of deterioration by secondary fermentation. Removal at the rate of 10 to 30 cm per day minimises losses, and a range of new ensiling techniques reduce exposure time and surface area (Park and Stronge 2005). Good ensilation achieves better fodder conservation than hay (Table 2) (Ernst *et al.* (1990).

Table 2. Typical dry matter losses (%) in silage and hay making.

Location	Silage (wilted 36 h)		Hay (6 days' drying)
	Round bale (wrapped)	Bunker	
Field	<5	<5	22
Storage	<5	10	5
Feeding out	3	3	1
Total	13	18	28

Silage leachate characteristics

The pollution potential or BOD of silage leachate is significant; the strength of the leachate from 1 t of silage with a moisture content of about 23% is equal to that of 18 000 L of sewage. The amount of leachate produced by maize silage depends on the moisture content of the crop and on the degree of consolidation.

Silage leachate has a low pH, is very corrosive to metal and will damage concrete. Any metal coming into contact should have a protective surface finish.

Grain and food-processing by-products

The dairy industry is drawing on an increasing diversity of grains and food-processing by-products, some of which liberate odours as a result of anaerobic decomposition and extended storage. Storage in a dry sheltered environment (in a shed or under a canopy) will avoid rainfall ingress, dust propagation and microbial decomposition. Grains and food-processing by-products should be stored on protected surfaces such as concrete pads or polyethylene or rubber mats, rather than in direct contact with soil. Drainage is essential. Feeding-out machinery should be dedicated to the purpose and not come into in contact with soil. Mycotoxins can contaminate some feedstuffs, so breathing apparatus is recommended. If contamination is found, the provenance of feeds should be traceable; soil shipped with fodder can introduce microbial pathogens to a farm.

Marc feeding and the use of by-products from the fruit and vegetable processing industries are contributing to the successful exploitation of waste from one industry to the benefit of another, but can aid the transfer of pests and diseases. Regulations control the transport of grapes and vines to avoid the spread of phylloxera; others govern the movement of fruit to avoid the export of fruit fly.

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5 Odour emissions and control

Odour refers to the aggregate effect of a mixture of gases on the sense of smell. For animal effluent and manure, it is the composite of over 170 trace compounds, including ammonia (NH_3), amines, hydrogen sulphide (H_2S), volatile fatty acids, mercaptans, alcohols, aldehydes, esters and carbonyls (Sweeten *et al.* 2006). Some of the compounds (e.g. H_2S and NH_3) have been monitored in detail individually owing to their impact on human and animal health (see chapter 6 'Occupational health and safety').

Odour is becoming an increasingly important issue as milking herds grow, notably in areas valued for rural residential blocks. The potential for odorous emissions to cause nuisance (inconvenience materially interfering with ordinary comfort) to neighbours cannot be dismissed. Although new developments may need to use odour assessment tools for decisions regarding siting, existing dairies must also understand odour generation and dispersion to implement effective odour control strategies.

Units of measurement—OU

Unfortunately, no individual component can be used as a marker to quantify livestock odour intensity; if so, the measurement and monitoring of odour would be a simpler matter than it is. Rather, the intensity of the odour must be measured by a trained human panel, a process referred to as olfactometry, which is described by AS 4323.3 'Stationary source emissions—Part 3: Determination of odour concentration by dynamic olfactometry' (Standards Australia 2001).

AS 4323.3 specifies the odour unit (OU) to report odour concentration. Odour concentration is measured by determining the dilution factor required to reach the detection threshold (the dilution at which the sample has a probability of 0.5 of being perceived). The odour concentration at the detection threshold is by definition 1 OU. Specific odour emission rates are expressed in units of $OU{\cdot}m^{-2}{\cdot}s^{-1}$.

Note that the European standard (CEN 1999), on which AS 4323.3 was based, uses units of $OU{\cdot}m^{-3}$ for concentration and therefore $OU{\cdot}m^{-2}{\cdot}s^{-1}$ for specific odour emission rate. Although both are correct, the units adopted by AS 4323.3 are more consistent with the definition of odour as a dilution factor.

Odour generation

Odour emissions are generated during the incomplete anaerobic decomposition of organic matter in manure. Area-based sources around the dairy include ponds, solids separation systems, manure stockpiles, feedpads, loafing paddocks and laneways. Silage, wet by-product storage and spilt feed are also significant sources of odour. The distribution of effluent and the desludging of ponds release odour, but the timing of such planned activities can be scheduled to minimise the impact on neighbours.

Anecdotal evidence suggests that, to date, most odour problems caused by dairies are a result of emissions from manure accumulated on laneways and feed areas (particularly after rain), spoilt grain and silage. The larger dairies more commonly have problems with odours from ponds than do smaller dairies.

Complaints about odours from the dairy itself may relate to a build-up of manure outside the washed areas (yard entry and exit, areas surrounding sump and solids trap) rather than the holding yard (as fresh manure is generally not considered to emit offensive odours). Where a dairy is close to neighbouring residences, the investigation of any complaint must consider whether noise (from plant and machinery, cows, radios etc.), dust or flies are at the root of the nuisance.

Odours could result from the reuse of treated effluent to wash yards, so options to improve the level of treatment before reuse must be considered (also see chapter 2.3 'Anaerobic, aerobic and facultative ponds').

Total odour emissions from a site are proportional to the surface area from which the odour is emitted (total odour emission equals the specific odour emission rate multiplied by the surface area). Therefore, for a given specific odour emission rate, the larger the area, the larger the total odour emissions. Intuitively, the volatile solids (VS) loading of the pond determines the specific odour emission rate, but limited data from the pig industry suggests that although higher loading rates may increase specific emission rates, the net result is that total odour emissions from the site may be reduced as a result of the smaller pond volume and surface area of the more heavily loaded pond (Skerman 2007). Hudson *et al.* (2004) determined that in piggery ponds with loading rates ranging from 53 to 454 g VS $m^{-3}\cdot day^{-1}$, a 350% increase in loading rate produced a 39% increase in odour emission in winter. It is noteworthy that seasonal variations in emissions were at least as great as those due to loading rate.

Skerman (2007) used this result to develop odour emission versus loading rate curves for different desludging periods in an effort to identify the loading rate that minimises pond odour emissions. For a 10-year desludging period, a loading rate of 180 to 240 g VS $m^{-3}\cdot day^{-1}$ minimised pond odour emissions in piggeries. The optimum loading rate was higher with more frequent desludging.

Similar efforts are required for the dairy industry, but a lack of emission rate data is currently a limitation. Other advantages of more heavily loaded ponds include lower construction costs and a less expensive cover if additional odour or greenhouse gas (GHG) control is needed. These opportunities warrant research aimed at identifying odour emission rates from ponds under a range of loading rates.

Solid–liquid separation traps also generate odour but their overall contribution is usually small owing to their limited surface area. However, sedimentation basins may be a more significant source, depending on their surface area, design and management. Beef feedlot research has found that emissions from sedimentation basins can exceed emissions from ponds if not managed successfully (Sweeten *et al.* 1977). This finding is relevant for some larger dairies and freestall operations.

Any surface where manure accumulates will generate odour if the moisture content of the manure exceeds around 70% (the point at which anaerobic conditions begin to prevail). Within beef feedlots, odour emissions from wet pads are commonly 25 to 100 times those from dry pads and peak 1 to 5 days after rainfall (Lunney and Lott 1995). During dry times, stock traffic may pulverise accumulated manure, producing 'fines' that are removed by the wind. Although dust is a pollutant in its own right, it will also cause odour when it comes into contact with the olfactory nerve.

Aside from moisture content, temperature is an important determinant of odour emission rates. That is, bacterial activity is more rapid under warm conditions than cool, with faster rates of decomposition and therefore odour generation.

Separation or buffer distance

The traditional regulatory approach to avoiding odour has been the imposition of separation or buffer distance requirements on new developments in an effort to allow the odour to disperse before reaching any potential receptor. Table 1 presents some commonly applied separation distances.

Table 1. Separation distances to nearest house (m).

	NSW	Vic	Tas	SA	WA
Pond	200	300	300	200	200
Solids trap				50	
Manure stockpile	200		300		
Land application	100	300 (slurry)	100[a]; 300[b]	100	100[a]; 300[b]

[a] Intermittent use only.
[b] Continuous use.

It could be argued that separation distance requirements should also be applied to the dairy and any feedpad. Both generate noise, light and, where manure is allowed to accumulate, odours. However, the determination of appropriate separation distance usually involves compromises; setback distances large enough to allow sufficient dispersion or attenuation under stable atmospheric conditions may unduly restrict new developments (or natural expansion), but small setback distances are insufficient to mitigate the frequency and severity of nuisance at some sites. It is unlikely that the separation distances from pond to receptor given in Table 1 are adequate for large operations, and farmers and regulators alike need additional tools to assist in planning. Fixed separation distances are not suitable for preventing nuisance impacts from odour as they are inflexible and unresponsive to site-specific issues (size and nature of operation, local weather patterns, topography).

Calculation of buffer distance

Some state guidelines for the pig and beef industries use empirical equations to calculate a suitable buffer distance according to the number of animals, site management practices, receptor type, local terrain and vegetation. The general nature of the equation is:

$$D = S \times \sqrt{N} \tag{1}$$

where

D = separation distance

S = composite site factor ($S_1 \times S_2 \times S_3 \times \ldots \times S_n$)

N = number of standard animal units.

In the dairy industry, N is the number of dairy cattle units adjusted for live weight and time on feedpad. As dairy developments in Victoria (the first state to implement this approach for dairies) are assessed against the requirements of the Victorian Code for Cattle Feedlots (Department of Agriculture Energy & Minerals Victoria 1995), a dairy-specific stocking intensity factor (S_1) was developed for feedpads in the Goulburn–Broken catchment (Dairy Cattle Feedpad Working Group 2002) to be used in calculating Equation 1 and achieve compatibility with the code.

The Dairy Cattle Feedpad Working Group (2002) based S_1 on field-trial-derived relationships between odour emission rates, stocking intensity and management of beef feedlots using model-predicted odour concentrations calibrated by receptor impacts. As no research data from dairy farms was available for corroboration, this approach is indicative only. Where the calculated separation distance is less than the minimum fixed separation distance (300 m), this minimum distance applies.

Although this approach overcomes some of the limitations of the fixed separation distance, it has not been rigorously tested, and its use is limited to dairy farms with a feedpad. The use of a site factor based only on feedpad stocking rate and management does not enable different scenarios for potentially the largest odour source—the effluent management system—to be considered (e.g. single pond vs. two ponds, solids trap vs. sedimentation basin).

The remaining site factors are obtained from Dairy Cattle Feedpad Working Group (2002).

International approaches to calculating separation distances

Although models developed in the USA and Europe are not generally applicable owing to the intensive nature of their dairy farming systems and differences in climate, the approach used to develop the OFFSET model in Minnesota, USA, warrants consideration. In the OFFSET methodology, all potential odour sources are listed and assigned a representative specific odour emission rate (an average derived from a comprehensive odour monitoring program) (Guo *et al.* (2005). The specific emission rates can be adjusted to account for any odour control measures such as permeable or impermeable covers. The total odour emission from the site is then determined by summing the contribution from each source (specific odour emission rate multiplied by the respective surface area).

Once the total odour emission is calculated, the setback distance is determined using standard curves with an ‘annoyance-free’ frequency from 91% to 99%. The curves were generated by dispersion modelling and calibrated by on-ground surveys (emission rates and receptor impacts were measured over 4 years in 85 farms). For the purposes of OFFSET, annoyance-free odours are defined as those odours with an intensity of <2 (defined as weak or mild odours that are not likely to be annoying) on the *n*-butanol odour intensity reference scale (0–5).

The advantage of the OFFSET approach is that it can account for the size and nature of a range of odour sources, including adjustments for odour control measures, and the frequency of impacts on receptors without site-specific modelling. Unfortunately, the Australian dairy industry covers a much broader range of climatic and topographical conditions than in Minnesota, and has differing regulatory targets, and would require the generation of at least regional standard curves. (Note, however, that OFFSET assumes that the receptor is always located downwind of the odour source in the prevailing wind direction, which is the worst-case scenario.) More critically, the availability of representative odour emission rate data from Australian dairy farms is extremely limited (see ‘Emissions data’ below).

Dispersion of odours

Although the complete elimination of odour is not possible, potential conflict with neighbours may be avoided with an understanding of the conditions that are more likely to cause odour plumes to travel long distances. Odour is carried away from the dairy by the prevailing wind. As it moves downwind, dispersion causes the odour concentration to decrease with increasing distance from the site. The rate of dispersion depend on atmospheric stability: a hot, windy day (unstable atmosphere) results in faster dispersion than a cold, calm, cloudless evening (stable atmosphere).

In some locations, odour plumes may travel long distances as a result of topographic features that confine plumes and limit their dispersion. Katabatic drift is the movement of cold air downslope within a valley, generally during times of stable atmospheric conditions. Farms in such situations are at additional risk of causing nuisance; new developments should avoid such sites.

Dispersion modelling

Computer-based odour concentration models can be used to determine the intensity and frequency of odours at specified locations around a source from local weather data. Ausplume is the dispersion model favoured by regulatory agencies around Australia; it should be used except where conditions make it unsuitable. Pacific Air and Environment (2003b) provides a guide to deciding which situations will be satisfactorily

treated by Ausplume, and proposes that the following situations require a more advanced model:

- ‘Critical receptors are located at a distance from the source that is greater than the minimum distance travelled by the plume in one hour, and there is evidence of significant effects of non-steady-state meteorology.
- More than one [piggery] is being considered in the same model application.’

Pacific Air and Environment (2003b) states that there are:

‘significant limitations to Ausplume, which need to be recognised and which might mean that use of another, more advanced, model is appropriate for a given situation. Another commonly used type of model is the Gaussian puff dispersion model, of which CALPUFF is the best-known example. For low-level emission sources such as piggeries, the differences between predictions from steady state and puff models are expected to be greatest for stable, near-calm (low wind) conditions, which generally lead to the highest predicted short-term concentrations.’

Meteorological data

Generally, a minimum of 12 months of hourly data is required as input to a dispersion model for a thorough assessment. The 12-month period selected must be representative of the normal range of conditions in the area. Pacific Air and Environment (2003a) reviews data requirements and lists potential data sources.

Emissions data

Unfortunately, there is little data describing odour emission rates from dairy farms. One research project (Feitz 2002, Wang and Feitz 2004) and three commercial investigations (Geolyse 2007, Holmes Air Science 2000, The Odour Unit 2005) provide the only Australian data available (Table 2).

There is clearly a need for additional information to explain the magnitude of the difference between the data sets, particularly within the three data sets derived from samples collected with an isolation flux hood. All samples were analysed according to the current Australian Standard or, in the case of Holmes Air Science (2000), its basis, CEN (1999). Discussions with the owner of the property sampled by the Odour Unit (2005) suggest that the anaerobic pond was not functioning effectively—a conclusion supported by the high COD and BOD results. Similarly, after 8 years of operation and sludge accumulation, Geolyse (2007) suggested that the performance of the treatment ponds was curtailed by lower hydraulic residence times, leading to increased odour emission rates. Differences in the treatment system design, age and maintenance history of ponds logically produce significant variations in measured odour emissions.

The wind tunnel measured data of (Feitz 2002) yield numbers between the other data sets. Most research shows that isolation flux hoods under-predict odour emissions relative to wind tunnels (Galvin (2005). Unfortunately, there is no fixed correlation between emission rates as measured by the different apparatus, so corroboration between data sets is not possible (Jiang and Kaye (1996). The debate regarding the most appropriate apparatus for odour sampling continues.

Table 2. Available odour emissions data.

Source of odour	Specific odour emission rate ($OU \cdot m^{-2} \cdot s^{-1}$)			
Reference	Holmes Air Science (2000) [a]	Odour Unit (2005) [b]	Geolyse (2007) [c]	Feitz (2002) [d]
Collection apparatus	Isolation flux hood	Isolation flux hood	Isolation flux hood	Wind tunnel
Anaerobic pond	0.38	1.35 (on dry crust) 82.7 (on wet crust) 75.4 (no crust)	6.4	8.1 (2–34)
Storage pond	0.12	2.12	9.0	
Sump and manure separator	1.48			
Manure stockpile		9.42	0.08	
Freestall pen	0.35			
Freestall channel	0.58		0.07 (flushed clean) 0.14 (dirty)	
Silage	7.90	916	0.31	

a: Large freestall operation, data collected over 2 days during fine autumn weather; maximum of two measurements reported; no loading rate information collected.
b: Medium-sized operation with feedpad; data collected during one fine and hot (32 °C) day in autumn; COD at anaerobic outlet 9400 $mg \cdot L^{-1}$; BOD 1900 $mg \cdot L^{-1}$.
c: The same freestall operation monitored by Holmes Air Science (2000) after approximately 8 years of operation, sampled during winter.
d: Monitoring at 30 ponds (primary and secondary) over 12 months on farms in Queensland, NSW and Victoria.

Three international studies identified odour emission rates on dairy farms:

- Mean odour emission of 4.7 $OU \cdot m^{-2} \cdot s^{-1}$ (range 0–10.3) from a single-cell earthen basin holding manure slurry (2.2% TS); sampling by isolation flux hood; olfactometry standard not identified (Zhao *et al.* 2007).
- 27 $OU \cdot m^{-2} \cdot s^{-1}$ from 3 single-cell earthen basins; 6.3 $OU \cdot m^{-2} \cdot s^{-1}$ from 1st cell of 4 multiple-cell systems, and 5.1 $OU \cdot m^{-2} \cdot s^{-1}$ from 2nd cell; sampling by wind tunnel in accordance with ASTM Standard E679–91 (Gay *et al.* 2003).
- 7–10 $OU \cdot m^{-2} \cdot s$ from 2 single-cell earthen basins and 2–3 $OU \cdot m^{-2} \cdot s$ from 2 freestall barns; sampling by wind tunnel to CEN (1999) (Bicudo *et al.* 2003).

Casey *et al.* (2006) compare emissions by animal types.

Hudson *et al.* (2004) identified significant spatial variability in emission rate (by as much as 10×) across the surface of piggery anaerobic ponds. They concluded that at least four odour samples are required to remove the uncertainty created by this spatial variability. Unfortunately, none of the sampling efforts listed in Table 2 meet that criterion.

The limited data available do not allow a representative range of odour emission rates to be selected. Therefore, the accuracy of any attempts at odour modelling for dairy developments must be considered questionable until additional information can be developed. A research program investigating emission rates from the range of sources around the dairy and the impacts of loading rates, age and maintenance regime on pond emissions is urgently needed.

Regulatory target criteria

The development of target odour criteria is complicated by the difficulties in odour sampling and measurement combined with a lack of suitable data on odour levels associated with annoyance and complaint (Galvin *et al.* 2007). In lieu of definitive information, state regulatory agencies have developed differing criteria (Table 3).

Table 3. Odour criteria summarised by Galvin et al. (2007) unless otherwise stated.

State	Percentile occurrence	Odour concentration	Averaging time	Assessment point
Queensland	99.5	0.5 OU point 2.5 OU non-point	1 h	Sensitive receptor (existing or future)
NSW	99.0	2 OU (pop. 2000+) to 7 OU (pop. 2)	1 s [b]	Sensitive receptor (existing or future)
Victoria	99.9	1 OU non-rural 5 OU rural + risk assessment	3 min	Property boundary
Tasmania [a]	99.5	2 OU	1 h	Property boundary
SA	99.9	2 OU (pop. 2000+) to 10 OU (pop. <12)	3 min	Sensitive receptor
WA [a]	99.5 99.9	2 OU 4 OU	3 min	Sensitive receptor (existing or future)

[a] Wang and Feitz (2004)
[b] A peak-to-mean factor (a conversion factor that adjusts mean dispersion-model predictions to the peak concentrations perceived by the human nose) must be applied to emissions before modelling .

If we take Victoria as an example, the 99.9th percentile 5-OU criterion means that the odour concentration at the point of interest has to be <5 OU for all but 9 h per year (0.1% of the time). The required buffer from site to receptor therefore increases with higher compliance frequencies and, where a population-based criterion is in place, with population density.

The different odour criteria adopted by states creates discrepancies across some regions. Wang and Feitz (2004) suggest that the Victorian odour criteria are significantly more restrictive than NSW, with a buffer distance 4.7 to 7 times larger. However, Galvin *et al.* (2007) suggest that although the criteria are different, modelling to Queensland, NSW and SA criteria produced similar buffer requirements for broiler farms in those three states.

Galvin *et al.* (2007) warn that the more stringent the percentile value is, the more likely that the modelled results fail to show the influence of terrain. That is, peak odour concentrations associated with atypical meteorological conditions dominate the results. Lower percentiles (99.5th to 98th) are more likely to filter out atypical conditions than the 99.9th percentile.

Wang and Feitz (2004) tried to define target criteria for the Australian dairy industry. They suggested 6.5 $OU{\cdot}m^{-3}$ (the recognition threshold for dairy odour), 1-h averaging and 99.5th percentile at receptor as appropriate criteria for the assessment of dairy farm odours, but based that conclusion on achieving an arbitrary separation distance of 500 m for the average emissions from the 9 farms modelled. Their work did not include any community survey or field panel assessment to verify the level of impact.

Odour control strategies

Proper siting, taking into account distances to neighbours, prevailing wind directions and topography, is the single most important factor in avoiding potential conflict. However, on existing farms, particularly those with sensitive receptors located close by, other strategies may be necessary to reduce emissions.

Attention to detail in general management and maintenance (good housekeeping) is an important factor in minimising complaints and maintaining good relationships with neighbours. Planning activities that are likely to release odours (desludging, solid and effluent application) for a time when odour impacts are less likely is also important.

Additional strategies such as chemically assisted solids separation, impermeable and permeable covers, partial aeration and odour control additives can control odours from ponds. Unfortunately, the costs of these strategies are significant and their efficacy varies. Fortunately, however, few operations currently need them.

Manure decomposing anaerobically (typically requiring a moisture content > 70%) is likely to emit odours. Management practices that eliminate any unnecessary accumulation of manure will help to reduce the potential for impacts on neighbours. Where it is not possible to eliminate those areas, regular and thorough maintenance is critical.

Feedpads, loafing pads and sacrifice paddocks

Feedpads should be designed to be cleaned regularly (preferably daily via flushing or scraping to minimise odours) and integrated with the effluent management system. However, the investment in large areas of concrete cannot be justified for all farms, and earthen feedpads and loafing areas are suitable provided they are designed properly (see chapter 4.2 ‘Feedpads, calving pads and loafing pads’).

In minimising odour emissions from earthen pads, the critical aspects include:

- a minimum slope of 2% to 4% to promote good drainage (Lunney and Lott 1995)
- regular maintenance to fill holes and maintain free drainage
- attention to seemingly minor details such as cleaning under fences to maintain drainage and removing manure that settles in collection drains.

Scrape earthen pads if more than 50 mm of manure has built up to reduce the manure load present during wet periods. Clean any earthen drains as often as needed to remove settled manure. If vegetation is established in drain beds, it will reduce flow velocity and trap manure; therefore, regular spraying with a broad-spectrum herbicide is necessary.

Spilt feed is particularly odorous if it becomes wet and spoils. Any feed accumulating behind feed bunks or around feedpads must be removed before it spoils.

Feed-out or sacrifice paddocks can be a significant source of odour following rainfall owing to the accumulation of manure and waste feed. Areas to be used for such practices should be selected to avoid affecting neighbours (consider buffer distance and prevailing winds) and rotated regularly to avoid an excessive (>50 mm) build-up of putrescible material.

Manure stockpiles

Water draining from stockpiled solids must be prevented from ponding around the pile, where it will maintain anaerobic conditions at the base of the pile. A compacted pad with a 2% to 3% slope to the effluent collection system is required for adequate drainage. Fill and compact any depressions made during manure removal.

If the manure stockpile is large or emitting odours, it may need to be windrowed and turned regularly until it dries enough to maintain aerobic conditions required for composting (see chapter 2.9 ‘Composting’). Such turning is likely to release significant odours and must be timed to avoid worsening the situation (see ‘Planned activities’ below).

Feed storage

Although the nature of odours from silage is different (often described as sweet or grassy) and usually less offensive than from manure, it may be of an intensity that causes complaints. The Odour Unit (2005) reported that odour emissions from the face of a particular silage bunker produced the one of the highest odour emission rates ever recorded (Table 2). Placing a cover over the disturbed face of the bunker may be necessary where neighbours experience effects. All leachate from the bunker must be captured and directed to the effluent collection system.

Operations that feed food-processing by-products (e.g. brewer's grain, cannery pulp) have an additional odour source. Although such odours may not be unpleasant, feed volumes stored on-site should be minimised as much as possible. Anecdotal reports of leachate from some cannery wastes inhibiting pond function (and increasing odour emissions) suggest that the waste's properties (moisture content, pH, EC) must be investigated; if they prove problematic, separate leachate storage may be required.

Spoiling grain can be a potent source of offensive odour and spills should be removed as frequently as they happen.

Dietary modification

Powell (2006) states that 'manure management should start at the front, rather than the back end of the animal.' Carbohydrates and proteins in manure are the two major energy sources for bacterial growth and odour production (Zhu *et al.* 1999), so strategies that reduce the amount of these constituents may reduce odour emissions from both the ponds and the other areas around the farm.

The principle odorous compounds resulting from manure decomposition include volatile fatty acids (VFAs), ammonia, amines, indoles, phenolics and volatile sulphur-containing compounds (Mackie *et al.* 1998). Zhu *et al*. (1999) confirmed that the most pungent odorous compounds (in pig effluent) originate from the decomposition of proteins. Sutton *et al.* (2006), however, suggest that reducing N output (and, by association, proteins) from cattle is challenging and limited by the ability to accurately formulate diets with the required nutrient availability (particularly where pasture comprises the major proportion of intake). In trying to reduce odours, take care not to diminish milk production and animal performance.

Miller and Varel (2001) suggest that VFAs from feedlot cattle are predominantly produced by the fermentation of carbohydrates, particularly starch. Improving the digestibility of grain supplements is one means of reducing waste starch output and therefore odour. Archibeque *et al*. (2006) established that feeding high-moisture ensiled maize plants reduced starch output in manure and the production of odorous compounds relative to dry rolled maize. In addition, Burkholder (2004) found that feeding steam-flaked maize to dairy cows increased N digestibility, reduced N output and reduce the rate of ammonia loss from manure and urine compared with dry rolled maize.

Although the variability in pasture-based systems precludes much of the opportunity for ration modification afforded to cows fed in freestall sheds and, to some extent, on feedpads, the formulation of supplementary feeds should be based on nutritional requirements to avoid overfeeding and reduce the excretion of undigested components.

Planned activities

Planned activities (cleaning solids traps, effluent irrigation, desludging, manure spreading etc.) should be timed to avoid effects on neighbours. Although these procedures will generate odour, the manager can select the timing of the activity to minimise emission impacts (Lunney and Lott 1995) by:

- avoiding timed activities if other emissions from other sources are high (for example, following rainfall), as odours are largely additive
- scheduling activities from Monday to Thursday to avoid operations immediately before the weekend
- performing operations in the morning to take advantage of warming conditions, which enhance dispersion, and to allow odour emissions to reduce before stable atmospheric conditions return with nightfall

- favouring days with unstable atmospheric conditions—warm, windy, little cloud cover and no rain forecast
- avoiding activities when the wind is blowing towards sensitive receptors.

Lunney and Lott (1995) report that odour emissions following manure spreading decay to negligible levels (~10% of initial emission rates) within 2 days. Excessive rates of application or uneven spreading may cause odour emissions to remain high for longer. Incorporation of manure as soon as possible after spreading not only reduces odour emissions, but also maximises the nutrient value of the manure by limiting N loss. If manure is not incorporated, follow-up rain may cause odour emissions to spike and result in nutrient loss in runoff.

Odours during irrigation of effluent may result from either the release of gaseous compounds or aerosol drift. If the latter is the cause, the nozzle and pressure combination of the irrigation equipment needs to be reviewed (low pressure, large nozzle size and low application height will minimise aerosol drift). The following measures may also help:

- Diluting the effluent with clean water may reduce odours during irrigation. Additionally, a short irrigation with clean water following the effluent may flush effluent from the large surface area of the vegetation.
- Pump effluent from the final pond within the treatment and storage system if possible. If there is only one pond, set the on a float to draw from under the surface and avoid sludge (but low enough to avoid air entrainment and floating material).
- Avoid application rates that lead to surface ponding.

Public relations

The importance of maintaining good public relations in reducing the perception of odours cannot be overstated. Informing neighbours before planned activities will not only avoid coinciding with any social events, but also help to retain goodwill through what should be only short-term impacts. If neighbours are experiencing an increase in odours, keeping the lines of communication open will allow the farmer to review the possible reasons and rectify problems before the neighbours feel their only option is a formal complaint to authorities.

Aesthetics and image are also important—a clean and well maintained farm will generate fewer odour complaints than a weed-covered, debris-laden farm. Plants may also help by providing a visual screen around the site (out of site is out of mind). Note, however, that although trees provide a windbreak that creates turbulence and vertical dispersion, they offer limited benefit during calm conditions, when odour plumes are most problematic (Lunney and Lott 1995).

Odour control strategies for ponds

Sweeten *et al*. (2006) suggest that problems caused by odours from ponds generally stem from overoptimism in design, performance, ease of maintenance and public tolerance of off-site impacts. They suggest that many problems could be avoided by not making the following mistakes:

- Designing to meet minimum guidelines (no capacity for natural expansion).
- Underestimation of organic loading rate (e.g. time on yard or feedpad, manure output per head).
- Inappropriate site selection.
- Attempting to accomplish both treatment and storage with one single-stage pond rather than multistage ponds (on larger farms).

- Insufficient sludge clean-out interval or failure to plan for sludge removal and use.

Where an effluent pond is established as being the cause of offensive odours, the following control strategies may help.

Chemically assisted solid–liquid separation

Solid–liquid separation may remove a portion of the VS before the anaerobic pond, but as most of the precursors to odour generation are contained in the finer particle fraction (typically <0.25 mm), which is not removed in solid–liquid separation, it has a limited capacity to reduce odour generation (Zhang and Lei 1998). If odour control and nutrient removal are the goals, chemically assisted solid–liquid separation may be required.

Chemical treatment involves the addition of coagulants and flocculants to alter the physical state of smaller suspended and colloidal solids and facilitate their removal by physical separation. Inorganic coagulants destabilise the net negative surface charge on the colloidal particles and promote the formation of flocs. Commonly used coagulants include calcium hydroxide (lime), ferric chloride and aluminium sulphate (alum). These metal ions also react with phosphate ions to form a precipitate and increase the removal of phosphorus from effluent.

Polymers promote flocculation or agglomeration of the flocs. Among commercially available natural and synthetic polymers, polyacrylamide (PAM) is the most common synthetic polymer, and chitosan is an example of a natural polymer with similar efficacy to PAM (Garcia *et al.* 2007). Timby *et al*. (2004) and Krumpelman *et al*. (2005) both confirmed that high-charge-density cationic PAM is suitable for dairy effluent. Polymers can be added separately or in combination with metal salts.

Garcia *et al.* (2007) recorded removal rates of up to 95% of total suspended solids (TSS) and 54% of total phosphorus (TP) from dairy effluent (3.2% TS) after flocculant (PAM) treatment and passage through a 0.25-mm screen. Zhang *et al*. (2006) found that gravity settlement of dairy effluent (3% TS) after the addition of the polyethylenimine (PEI) reduced TS by up to 58% and TP by 77%. Chastain *et al.* (2001) found that mechanical screening (1.6-mm) followed by treatment with PAM and 60 min settling time reduced TS, VS, N and P by similar amounts as solid–liquid separation followed by treatment lagoon.

Researchers investigating coagulants generally list ferric chloride and alum as more effective than lime (note, however, that lime is significantly less expensive). Barrow *et al*. (1997) identified removal rates of 89% of TS and 88% of TP from 1% TS dairy effluent following treatment with ferric chloride and 20 min of settling. Karthikeyan *et al*. (2002) noted that the removal of total solids from 1.6% TS dairy effluent following 30 min gravity settlement improved from 30% without coagulants to 65% with alum and 70% with ferric chloride. Kirk *et al*. (2003) also demonstrated significant improvement following the addition of coagulants and favoured the use of alum over ferric chloride owing to its slightly better performance and price.

Generally, combinations of inorganic coagulants and PAM remove more TS and P than either alone (Krumpelman *et al.* 2005, Timby *et al.* 2004, Zhang *et al.* 2006).

Although the referenced papers identify the dosage rates used, these must be followed cautiously as the required dosage rates increase with increasing TS concentration of the effluent. Apart from wasting chemicals, adding excessive amounts can actually impede solid–liquid separation owing to destabilisation of flocs (Zhang *et al.* 2006). A bench-scale test using a sample of the effluent is required first to identify the most suitable chemical and optimum dosage rate.

The use of chemically assisted solid–liquid separation is limited for the following reasons:

- Coagulants require specialised equipment (and significant capital cost) to achieve the rapid mixing necessary for maximum performance (Metcalf & Eddy Inc. 2003).
- Monitoring short-term changes in effluent concentrations and responding with changes in dosage rate is difficult (Worley *et al.* 2005).
- Chemical costs are significant, ranging from US$56–80 per cow y^{-1} (Sherman *et al.* 2000) to US$104 per cow y^{-1} (Barrow *et al.* 1997).
- Higher dosage rates of coagulants and flocculants reduce pH, in turn affecting other treatment processes (e.g. anaerobic digestion).
- Some flocs are too delicate to be removed by screening (Barrow *et al.* 1997) and require sedimentation for removal.

As separated solids typically have a moisture content > 80%, stored solids will quickly become anaerobic and produce an additional odour source (negating some the improvement sought) unless handled properly (see 'Manure stockpiles' above). Screw presses and centrifuges are among the few devices able to achieve the <70% moisture content necessary to avoid anaerobic conditions in separated solids (see chapter 2.1 'Solid liquid separation systems').

Impermeable covers with gas collection

Impermeable covers are designed to trap all gases produced during decomposition. In addition to removing a significant areal odour source, they allow the captured gas to be flared or used as an energy source, both resulting in combustion of any odorous compounds (see chapter 8.1 'Production and beneficial use of methane'). Flares typically destroy >95% of volatile organic compounds, which is generally sufficient for odour control, but high-temperature catalytic or thermal incineration may be required for complete odour destruction.

An alternative to combustion is to pass the collected gases through a biofilter, in which microorganisms reduce the organic compounds to less offensive forms. A basic biofilter comprising a 300-mm-deep mix of straw and compost over a chamber formed by shipping pallets reduced odour emissions from ventilated pig sheds by 82% and hydrogen sulphide by 80% (Nicolai and Janni 2001).

Permeable covers

Permeable covers reduce emissions by acting as both a partial barrier (resistance to mass transfer) and as a biofilter providing surface area for biological treatment (Regmi and Surampalli 2007). Biofilters provide a suitable environment for aerobic microorganisms to oxidise odorous gases and reduce odour emissions.

Permeable covers may be made from materials such as supported straw, geotextile, vegetable oil or clay balls. Fortunately, these covers are often unnecessary for dairy anaerobic ponds, as the nature of the effluent commonly results in the natural formation of a crust (see chapter 2.3 'Anaerobic, aerobic and facultative ponds'). Bicudo *et al.* (2001) measured odour emissions from a crusted swine manure storage over 5 months and determined a mean emission rate of 7.3 $OU \cdot m^{-2} \cdot s^{-1}$ with crust (in early spring and autumn) and 13.6 $OU \cdot m^{-2} \cdot s^{-1}$ without crust (in summer). Misselbrook *et al.* (2005) suggest that crusts reduce ammonia emissions from dairy slurry stores by approximately 50% but did not measure the impact on odour. Monitoring at an Australian dairy by the Odour Unit (2005) showed a 98% reduction in odour emissions from a dry crust relative to the liquid surface (Table 2).

Crusts, therefore, are beneficial as an odour control strategy and should be left intact if they are not causing problems (see chapter 2.3 'Anaerobic, aerobic and facultative ponds'). However, two issues that may affect odour emissions require further research:

- Wet, recently formed crusts appear to increase odour emission rates (Bicudo *et al.* 2001). Does rainfall cause a similar outcome and, if so, over what period?
- Some gas concentrations appear to increase in the liquid (Bicudo *et al.* 2001). What impact will this have on emissions from storage ponds or during effluent distribution?

In ponds where a natural crust cannot be generated, artificial permeable covers may be necessary to control odours. In their literature review, Hudson *et al*. (2006a) provide a comprehensive summary of research into cover efficacy, and including laboratory-scale reductions in odour emission rates of 71% to 84% from piggery effluent. Subsequent field trials (Hudson *et al.* 2006b) found reductions of 87% to 90% by both supported straw covers and a supported geotextile (a double layer of polypropylene weed mat). Installation costs ranged from A$7.50 m^{-2} for the geotextile cover to A$12.00 m^{-2} for supported straw covers.

In US-based research, Clanton *et al*. (2001) suggested that unsupported straw covers require a thickness of up to 300 mm to keep straw afloat and dry enough to act as a biofilter. Husdon *et al*. (2006b), however, suggested that supported straw covers were effective even when the straw had undergone significant decomposition and the thickness had decreased to 20 mm.

Hudson et al. (2007) reported on the long-term efficacy of three cover types over 3 years at piggeries in Queensland. Average odour emission rates were reduced by 76% by a polypropylene cover overlain by shadecloth (for UV protection), 69% by shadecloth only, and 66% by a supported straw cover. Research by Regmi and Surampalli (2007) supports the suggestion that geotextile fabric covers are as effective as straw covers.

Partial aeration

Owing to the different biological pathways involved, aerobic treatment emits little odour compared with anaerobic treatment. As naturally aerobic ponds are not a practical option for agricultural effluent treatment, mechanical aeration is sometimes used to achieve aerobic conditions in a much smaller pond than would otherwise be necessary. However, complete stabilisation via mechanical aeration is not normally economically justifiable, as the power requirement for maintaining a completely mixed state is very high—15 to 30 $kW{\cdot}ML^{-1}$ for mechanical aerators and 10 to 30 $m^3{\cdot}ML^{-1}{\cdot}min^{-1}$ for diffused-air devices (Metcalf & Eddy Inc. 2003).

Partially aerated or ‘stratified’ ponds (where only the surface layer is aerated) have been investigated for odour control (Westerman and Zhang 1997). The aeration level recommended varies from 33% to 50% of the daily BOD load (Vanderholm 1984) and up to 50% of the daily COD load (Barker *et al.* 1980). For dairy effluent with a COD:BOD ratio of 6.9 (ASAE 1999), the latter recommendation is 7 to 10 times the former. As Vanderholm’s recommendation was based on domestic effluent (where the 5-day BOD test represents a greater proportion of the ultimate BOD than in dairy effluent), the more conservative recommendation of Barker *et al*. (1980) should be used for design purposes.

At a rate of oxygen transfer by a typical mechanical aerator of no more than 1.0 kg O_2 kWh^{-1} (Cumby 1987, Metcalf & Eddy Inc. 2003), the energy consumption required to meet 50% of the daily COD load would be at least 120 kWh per cow per year, assuming that 10% of the 0.65 kg per cow COD output is collected (see chapter 1.2 ‘Characteristics of effluent and manure’).

Zhang *et al.* (1997) suggest that continuously supplying sufficient oxygen to maintain a dissolved oxygen (DO) concentration of 0.5 $mg{\cdot}L^{-1}$ in a surface layer of 0.15–0.3 m depth appears to offer acceptable odour control. A deeper aerated layer (or higher DO concentration) is necessary where aeration ceases for more than 9 h. Ginnivan (1983) recommended similarly shallow depths (0.08–0.4 m), and Barker *et al*. (1980) recommended a slightly deeper layer of 0.6 m. Practical constraints in aerator design

and function may limit how shallow the depth of the aerated layer can be (Westerman and Zhang 1997).

Odour control additives

Commercial products marketed as a solution to odour problems continue to attract interest despite a lack of scientific evidence of their efficacy. However, some operators believe them to be somewhat effective.

The many odour control additives available can be grouped according to their method of action:

- Digestive agents—additives that presumably alter the microbial community to enhance the degradation of odorous compounds or reduce their production.
- Masking agents—volatile oils with a stronger but more acceptable odour than the nuisance odour.
- Counteractants—aromatic oils that neutralise an odour. However, owing to the complex nature of odour, it is unlikely that a single counteractant could be effective (Lunney and Lott 1995).
- Disinfectants—germicides that alter or eliminate bacterial action. Such chemicals are often toxic and therefore impractical, as well as expensive.
- Oxidising agents—chemicals that oxidise odorous compounds (and may also provide some disinfection), including ozone.
- Bio-catalysts—bacteria and enzymes that encourage the formation of non-odorous end products such as methane rather than the by-products of incomplete digestion.
- pH modifiers (particularly for ammonia control).
- Adsorbents—commonly zeolite or sphagnum peat, which perform a similar function to biofilters (see 'Impermeable covers with gas collection' above).

McCrory and Hobbs (2001) provide a comprehensive review of additives, their various modes of action and their efficacy. FSA Environmental (1999) include a comprehensive listing of research papers with detailed information on odour control additives.

Research shows that bio-catalysts may offer some scope for reducing the time needed for establishing a suitable population of bacteria in anaerobic ponds during startup (Dugba and Schneider 2000). However, anecdotal evidence suggests that seeding a new pond with sludge from an operational pond will have a similar effect (see chapter 2.3 'Anaerobic, aerobic and facultative ponds').

The pH modifiers also demonstrate reliable efficacy. In this case, the pathway for the emission of ammonia is well understood and predictable. As pH controls the equilibrium between ammonia (NH_3) and ammonium (NH_4^+) in solution, pH modification may reduce ammonia volatilisation. Of the pH modifiers, acids have been shown to be consistently effective (but can be expensive or hazardous and corrosive), but base-precipitating salts offer only a short-term effect and must be reapplied frequently (McCrory and Hobbs 2001). Unfortunately, ammonia is not well correlated with odour, so the impact may not be significant.

Commercial additives containing saponins appear to offer some scope for conserving ammonium, but the mechanism by which this is achieved is unclear. Further, research results have not been consistent; for example, Andersson (1994) found that the cost of additive outweighed the value of the N retained in slurry.

Sweeten *et al*. (2006) summarise work by Purdue University to evaluate 35 digestive agents claimed to reduce odour. Each additive was tested in a simulated manure pit over three 42-day periods and compared with four untreated controls. At the 95%

confidence level, 7 of the 35 additives reduced hydrogen sulphide levels but none reduced odour emissions as measured by olfactometry.

In local research, Nick Bullock & Associates (2007) report on four concurrent demonstration trials using probiotics in conjunction with low-energy aeration of dairy effluent ponds. Unfortunately, the results were not conclusive, as BOD, VS and nutrient levels were not significantly reduced by the treatment. Occasional increases in odour levels were observed when aeration disturbed the settled sludge. Problems with blocked air stones were a continuing problem in most of the ponds. At the field scale, it is difficult to reconcile that tens of litres of additive can have a significant impact on a pond containing (typically) millions of litres of effluent and billons of bacteria. In addition, as odorous compounds are produced via many different, and as yet largely unresearched, pathways, one product is not likely to work in all situations. Researchers have been unable to establish reliable guidance on which odour control additives are effective under what conditions. The most common conclusion is that the use of additives is an unreliable and potentially expensive option for odour control.

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6 Occupational health and safety

Common sense is a necessary ingredient to avoiding accidents on farms. However, everyone can make mistakes for various reasons, including time pressures, stress, tiredness, and working with unfamiliar or poorly maintained equipment. Therefore, effluent system designers must consider occupational health and safety (OH&S) issues to avoid introducing unnecessary risks.

Several fatalities on dairy farms have resulted from accidents while effluent systems were being maintained. In one instance, a man drowned when the tractor he was using to agitate an effluent pond slid into the pond and sank (WorkSafe Victoria nd.). In another instance, five people died when one man entered a manure pit to repair an agitator shaft; he and his four rescuers all died of asphyxiation.

General OH&S issues on dairy farms

Dairy-specific publications such as *Dairy Safety: A Practical Guide* (WorkSafe Victoria 2006) help farm owners to provide a safe work place. *Dairy Safety* applies a three-step process (find the hazards, assess the risks, fix the hazards) to on-farm activities and provides a good summary of potential hazards. Additional information for farmers is available at http://www.dairysafety.org.au/.

Effluent systems pose unique hazards, as routinely generated gases pose a safety threat in confined spaces. A confined space is any enclosed or partially enclosed structure (e.g. vat, tank, pit, pipe, silo, container, reaction vessel, receptacle, underground sewer, shaft, well, trench, tunnel) if the space has:

- restricted entry or exit
- hazardous atmosphere
- a risk of engulfment.

Manure gases

Storage, pumping, mixing, spreading and cleaning-out can release large amounts of gases from decomposing manure. There are four gases of primary concern:

- Hydrogen sulphide is a highly toxic gas that is heavier than air. It can cause dizziness, unconsciousness and death. At low concentrations it smells like rotten eggs, but at higher concentrations it deadens the sense of smell, and no odour can be detected.
- Carbon dioxide is an odourless, tasteless gas that is heavier than air. It displaces oxygen in confined spaces, which can result in asphyxiation.
- Ammonia is lighter than air. It has a pungent smell and can irritate the eyes and respiratory tract.
- Methane is also lighter than air. The main hazard is explosion within flammable limits (5%–15% CH_4). Explosive concentrations can occur during agitation or when the gas is trapped in an improperly ventilated space (see chapter 8.1 ‘Production and beneficial use of methane’). As methane is odourless, you will not be able to detect dangerous situations by smell.

ASAE (2005) and Schiffman *et al*. (2006) contain information on the characteristics of these gases and the concentrations at which they pose a hazard.

System design to minimise hazards

Consideration of likely hazards and risks at the design stage can prevent or at least minimise the hazards during subsequent system operation. This is particularly important with confined spaces. Refer to state-based confined-space regulations for further information on situations that constitute a confined space and the duties of designers in such situations.

Under state OH&S acts, designers are required to consider safety in their designs. Where a confined space cannot be avoided, designers are required to either eliminate the need to enter the space or reduce as far as practicable the need to enter it. For example, manure solids will settle in pumped sumps if held for more than 30 min, so some form of mechanical or hydraulic agitation is required (and has traditionally been installed) to obviate the need for the operator to enter the pit to remove solids. A thorough risk assessment at the design stage should highlight the need for such pumps and agitators to be easily removed rather than require someone enter the pit to perform routine maintenance.

System designers should also seek to eliminate the need to use tractors near the edge of effluent ponds (e.g. for agitating or pumping). Where this is not possible, safe systems of work should be specified and adopted, such as using low barriers or chocks to prevent the tractor from moving backwards. A suitable barrier would be required at every access point (see chapter 2.8 ‘Desludging and pond closure’).

All design plans should include a description of safety procedures for management and maintenance specific to the farm. Plans should include a statement reminding farmers of the need to adopt safe working practices for activities involving dairy effluent. Safe work practices are based on ensuring that workers have appropriate training for the task, have completed a Job Safety Analysis before starting the task, and receive adequate supervision while completing the task. A general list of risks and control measures is provided below.

Checklist of risk controls

The following checklist is a compilation of points raised by state-based guidelines (DairyCatch 2006, McDonald 2006, NSW Dairy Effluent Subcommittee 1999) with additional information from (WorkSafe Victoria 2006). The list focuses on effluent-specific issues and does not preclude any other requirements (e.g. guards to be fitted, electrical work to be carried out only by a qualified electrician).

Collection and conveyance

- Where pit and platform wash is piped directly to the effluent pond, install a water seal or gas trap to prevent gases from entering the dairy.
- Floodwash tanks must be installed on stable foundations and supports. Supply engineering computations or drawings certified by a structural engineer before construction.
- Children and inexperienced staff must not handle hydrant wash systems, as the high pressures and resulting forces can cause the nozzle to swing wildly.
- Sumps and solids traps must be covered or surrounded by fencing (including a lockable gate) to exclude children and stock. In some cases, standard swimming pool fencing could be used.
- Observe the requirements of confined-space regulations.
- Do not enter manure pits without a respirator and an emergency plan. An observer who understands safe rescue procedures should supervise any manure pit work.

- Do not smoke, weld, grind or use an open flame in a poorly ventilated area.
- Ensure that any exposed moving part on an effluent pump is guarded.

Storage and treatment

- Effluent ponds can form a substantial crust that supports subsequent weed growth. Although the crust may look like solid ground, it may not support the weight of a person or animal. Fence ponds immediately after construction to exclude children and stock.
- A warning sign must be mounted on the fence near the entry gates saying ‘Danger—Manure Storage’.
- Locate fences a sufficient distance from banks to allow machinery access around the toe of the batter.
- Eliminate the need to use tractors near the edge of effluent ponds where possible. If this is not possible, use barriers or chocks to prevent the tractor from moving backwards.
- Machinery may collapse unstable or narrow embankments. See chapter 2.5 ‘Pond design and construction’ for appropriate batter slopes and embankment widths. Investigate and rectify any evidence of slumping or undercutting, or embankments may collapse.
- Maintenance and desludging operations require extreme caution, as clay surfaces can become slippery when wet. Topping the embankment with gravel (at least to designated access points) will help maintain vehicle and pedestrian traction.
- Avoid the frequent use of pond embankments as laneways unless additional width and gravel surfacing are provided.
- Place a rescue rope and float within the fenced-off area around the pond.
- Owing to the risks of gas ignition and explosion, a specialised safety plan is required for any farm with a covered pond or other digester.

At any time

- All farm machinery must be regularly maintained according to manufacturers’ instructions, and all controls must be clearly marked. Do not use faulty machinery.
- Observe appropriate hygiene practices: no smoking, eating or drinking around the dairy; wash hand following contact with effluent and manure.
- Provide appropriate clothing and protective equipment such as gloves, aprons, rubber boots, goggles and other skin protection, and ensure that it is worn by staff who come in contact with animal effluent and manure.
- Maintain or replace all personal protective equipment regularly.
- Avoid inhalation of aerosols during reuse of effluent for yard or alley washing or spray irrigation.
- Follow effluent and manure management guidelines; poor practices increase the health risks associated with flies and insects.
- Supervise children visiting the dairy.
- Whenever chemicals are used, read and understand the Material Safety Data Sheet for the chemical involved and follow the safety precautions prescribed.
- Vaccinate people for Q fever. Vaccinate livestock for leptospirosis.

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7 Monitoring and sampling

As part of ongoing farm management, it is prudent to regularly monitor a range of aspects related to the effluent management system. The results of this monitoring can provide feedback to management and support decisions relating to system optimisation, including maximising enterprise production and viability while minimising risks to the environment, stock health and human health. In addition, the results of regular monitoring should enable management to fulfil any obligations under regulatory authority requirements.

A detailed account of the requirements of sampling and monitoring for intensive livestock industries is provided by Redding (2003); that document should be used as a basis for the parameters to monitor and the procedures for sampling. However, the document was prepared mainly for intensive livestock systems where effluent was being applied to cultivated land, not to intensively grazed pastures such as those used for dairy production. More research is needed to establish acceptable levels of monitoring for the dairy industry. The use of overall farm monitoring such as that outlined in chapter 3.1 ‘Nutrient budgeting’ and the use of a nutrient management plan or the Farm Nutrient Loss Index (FNLI) (Gourley *et al.* 2007) are recommended.

A simple but important and often overlooked aspect of monitoring is the recording of the location and timing of sampling and the appropriate storage of results.

What to monitor?

Base parameters to monitor

The actual monitoring required may differ from that detailed by Redding (2003), depending on the type of effluent management system used, the risks involved, strategies developed to minimise these risks, and specific recommendations from appropriate guidelines such as Tasmania’s State Dairy Effluent Working Group (1997), or regulatory authority requirements such as the relevant Environment Protection Act or State Environment Protection Policy (for example, SA’s Environment Protection Authority (2003)). This section presents the most comprehensive list possible of parameters to monitor in an attempt to cover most situations, so not all of the parameters listed may have to be monitored on any one particular enterprise.

Recording production information and environmental variables

Typical dairy management will include regular monitoring and recording of general farm management procedures, inputs and production levels. This practice will provide important information for assessing the results of effluent system monitoring. Keep records of any management changes that are likely to change production levels or effluent quality, along with details of:

- farm infrastructure—water storage and drainage system levels, surface runoff quantity and quality, pumping volumes
- pasture and crop monitoring—plant growth rates, plant symptoms (e.g. of salting or soil nutritional imbalances), crop yields or production levels, applications of fertiliser or soil ameliorants; irrigation quantity and water quality; a paddock walk is a useful way to identify these things
- stock performance—stocking rates, stock health, milk quantity and quality
- public amenity—to maintain a certain level of public and neighbour amenity.

Environmental variables should also be recorded as part of dairy management. Rainfall, weather (evaporation, wind, temperature etc.), groundwater levels and drainage line flows are all important parameters to record.

Monitoring of storage and treatment lagoons

The parameters that require monitoring in effluent storage and treatment processes are described in detail in chapter 2.3 ‘Anaerobic, aerobic and facultative ponds’.

Monitoring of effluent reuse area

- Soils—The intensity of monitoring will vary. Divide areas treated with effluent (both solids and liquids) into management areas, segregating those areas that have significantly differing management, soil type, crop or pasture, stocking rate, irrigation or effluent application rate. Each management area should then have at least one soil monitoring point on a representative area. Topsoil and subsoil samples should be analysed annually to assess for nutrient deficiencies or excesses. Take samples are selected from the same monitoring points over time for comparison.
- Groundwater—Install groundwater monitoring bores (or piezometers) within and next to reuse areas. Determine the number and location of bores in association with an assessment of local shallow hydrogeological conditions. These bores should be used to monitor fluctuations in shallow (0–3 m) groundwater levels and groundwater salinity content twice a year. A simple piece of slotted PVC pipe makes an effective monitoring bore. A simple hand-held EC meter will facilitate regular, easy, cheap assessment of water salinity levels. Assess variations in groundwater level and quality in conjunction with seasonal conditions and effluent application practices.
- Surface water—Collect any excess surface water from the reuse area for reuse and divert uncontaminated rainfall runoff to local natural drainage lines. Monitor surface runoff for quantity and quality.

Example parameters to be monitored

An example of some of the variables that may require monitoring as part of an effluent management system are listed in Tables 1 and 2. These tables provide the most comprehensive list possible of parameters that may need to be monitored. The degree of monitoring on any particular site will need to be assessed on an individual basis considering all effluent system facets and management. The list of parameters that will require monitoring will vary considerably from site to site.

Table 1. Comprehensive list of parameters that might have to be monitored (not all of these will be required on any one site).

Parameter	Specifics	Frequency	Timing
Effluent	Effluent generated	monthly	
	Effluent lagoon water quality	monthly	
	Salinity (hand-held meter)	weekly	
	Additional water onto farm	monthly	
	Dilution rates	at irrigation	
	Salt & nutrient loadings	annually	
	Water budgeting information	bi-annually	Feb & Oct
Climate	Rainfall	daily	
	Evaporation	weekly	
	Growing conditons (wind, temp etc.)	weekly	
Irrigation	Irrigation applications	at irrigation	
	Area irrigated	at irrigation	
	Reuse sump capacity	at irrigation	
	Waterlogging	continual	
	Salinity (surfae evidence)	continual	
	Surface ponding	continual	
	Soil moisture	continual	
Soil physics	Channel or drain erosion	continual	
	Physical deterioration (slaking etc.)	continual	
Soil chemistry	General	twice a year	Mar & Oct
	Detailed	every 2 years	March
	Salinity (hand-held meter)	as required	
Crop production	Crop health	continual	
	Production—product removed	as occurs	
	Product nutrient levels	annually	Mid season
	Leaf analysis	as required	
	Fertiliser and gypsum rates	as applied	
	Leaf analysis	as required	
Runoff	Off site runoff—quanitity & quality	as occurs	
Groundwater	Depth to water table	twice a year	Mar & Oct
	Chemical parameters	twice a year	Mar & Oct
Native vegetation	Tree health—mature trees on site	continual	
	Tree health—adjoining stands	continual	
	Tree health & mortality—plantations	continual	
Other	Record contractors onto farm	as occurs	

Table 2. Technical parameters that may need monitoring in a dairy effluent management system.

Characteristic	Parameters	Characteristic	Parameters
Wastewater	pH	**Soils – detailed**	Aluminium
	Total nitrogen		Boron
	Total phosphorus		Copper
	Total dissolved solids		Zinc
	Biological oxygen demand		Iron
	Salinity (EC)		Sulphur
	Na		Manganese
	Ca	**Soils – physical**	Slaking
	Mg		Dispersion
	K		Cracking
	Cl		Moisture
	Sodium adsorption ratio		Pugging
Soils – general	pH (H_2O)		Erosion
	pH ($CaCl_2$)		Waterlogging
	Salinity ($EC_{1:5}$)	**Groundwater**	EC
	Chloride		Chloride
	Organic carbon		Calcium
	Total nitrogen		Potassium
	Olsen phosphorus		pH
	Total phosphorus		Total nitrogen
	Exchangeable Na		Sodium
	Exchangeable K		Magnesium
	Exchangeable Ca		Total dissolved solids
	Exchangeable Mg		Phosphorus
	Ca:Mg		Depth
	CEC	**Crop**	Nitrogen
	Exchangeable sodium %		Phosphorus
	Skene potassium		Potassium

Instrumentation

Wireless sensor technology for feedback and control systems, GPS guidance systems, GIS mapping techniques and computer-based farm planning tools allow more precise measurement, support record keeping and reduce labour on the farm. Dedicated instruments can be used to measure important parameters of dairy waste management systems. The purpose of instrumentation is to:

- collect data for comparison with predicted or modelled outcomes
- maintain records for regulatory purposes and adjustment of systems to improve performance
- yield information for research and innovation.

Instruments are available to measure and record feed and water use, rainfall and evaporation, stock movement patterns, milking times, disease incidence, milk volumes and chemical usage, much of it at regular time intervals. Even basic instruments such as water meters, thermometers and pressure gauges can be handy.

Monitoring effluent

Water use and effluent conveyance can be monitored to reveal volumetric flow rate, power consumed, pipe and pump pressure, and volumes. Ultrasonic meters, magnetic flow meters and pressure gauges are by far the most common devices installed for

effluent monitoring. The recording of information for subsequent analysis when time is available is important.

Load cells, depth gauges and level sensors can be used to control discharge from pump sumps to stabilisation ponds. Float sensors are commonly used to control sump pumps and ponds and can be used to actuate surface aerators. Effluent quality sensors are uncommon. pH and chemical conductivity gauges can be used, but rarely are nutrient or dissolved oxygen sensors used.

Paddock monitoring

Sites on which manure and effluent are applied can be monitored. Although sap flow meters and leaf turgor sensors are available, by far the most common instruments are soil moisture meters such as tensiometers, ceramic and gypsum blocks, capacitance probes, neutron moisture probes, and time domain reflectometry meters.

A key objective is to capture spatial characteristics. Remote sensing devices can be used to aid this objective by monitoring temperature and canopy radiation from suspended points, satellites or high vantage points.

The range of devices available and their ease of use, accuracy and compatibility with computers, mobile phones and radio networks is extensive. As new technologies prove their worth, the take-up rate will increase and the unit cost will decrease.

Examples of suitable instruments

Rather than try and measure a host of variables with a range of specialist instruments, experience indicates that the simpler the instrumentation, the more robust will be the data. For example, rather than monitor effluent discharge, it is easier to use an off-the-shelf water meter to record water consumption; if all the water used becomes effluent, the results will be reliable. If necessary, a rain gauge and accurate surface area computations will accurately indicate additional runoff volumes. The characteristics of the effluent and changes in its characteristics over time can be gauged with an off-the-shelf conductivity meter. Very common instruments include water meters, pressure gauges, level sensors and soil moisture sensors. Equipment quality can usually be gauged by price. Although the accuracy of instruments for measuring effluent parameters can be dubious, case studies confirm that relative performance is critical: reliability and consistency of reading are usually of greater significance than accuracy.

Table 3 lists instruments that have well-established utility for dairy effluent use.

Table 3. Potentially useful instruments.

Parameter	Instrument
Flow rate	Bucket & stop watch, volumetric tank, load cell & stop watch, magnetic flow meter, weir plate & flume
Change in concentration	Conductivity meter, turbidity gauge, flask & stop watch, colorimetric test
Rainfall	Ramped gauge
Evaporation	Class A pan
Wind speed and direction	Anemometer
Soil moisture	Gypsum or ceramic block, neutron moisture probe, tensiometer, time domain reflectometry probe, capacitance probe, SENTEC meter, gravimetric sampler
Soil nutrient status	pH kit, hydrogen peroxide test, hydrometer

Although flow meters, hand-held pH and EC meters, level sensors, pressure gauges and visual observations are commonly used, more specialised instruments are under study. Kizil (2006) documents the evaluation of a gas sensor to estimate the nitrogen, phosphorus, potassium and ammonium content of cattle manure. Van Kessel and

Reeves (2000) evaluated cheap and simple techniques for assessing nitrogen levels in manure, using hydrometers, EC meters, colorimetry and a hand-held nitrogen gas meter.

The hydrometer proved to be the only quick test available for measuring total N in effluent. Singh and Bicudo (2004) found that a conductivity meter and conductivity pens gave more reliable results for ammonium than hydrometers. Provolo and Martinez-Suller (2007) demonstrated the value of EC for assessing total Kjeldahl nitrogen (TKN) and total ammoniacal nitrogen (TAN), the limited value of using EC to determine potassium, and no value at all for phosphorus (P) assessment. Lugo-Ospina et al. (2005) studied quick tests for P in dairy manure. Both dissolved P and total P could be determined reasonably accurately with a hand-held reflectometer in association with specific gravity tests.

To reduce the cost of analytical tests, instruments can be used to determine solids content and infer P and N levels. Higgins et al. (2004) found that knowledge of animal growth stage (which can be measured) and solids content could be used as proxies to predict the total N and total P contents of liquid animal manure.

The use of quick tests must be investigated under Australian conditions, as the quality of surface water and groundwater sources is variable (particularly EC) and may alter results.

Assessing results and reviewing performance

An annual revision of the performance of the dairy enterprise and of the effluent management system is recommended. Assess the preformance of the effluent management system in conjunction with the results from the monitoring of all site environmental, management and production parameters. Some results may need to be assessed by suitably qualified personal.

Record the results of all monitoring and store them to ensure that they are readily accessible. It is also important to:

- assess management variations proposed or instigated in response to monitoring results
- review the effectiveness of the monitoring process and any variations that are required.

System selection issues

The accuracy and frequency of the required measurements and the cost must be considered in equipment selection. Sensors must be compatible with coupling and output devices as well as with other sensors being used. The user should have experience with the technology.

Calibration

Calibration of any system is necessary. Blind reliance on numbers can be misleading. Instruments are best calibrated by applying a range of known static conditions to the sensor. Where this is not feasible, test each component of the instrumentation system; equipment suppliers will often do this.

Often it is necessary to develop another measurement system of greater precision to act as a standard for comparison of the main system. The data collected in calibration experiments must be analysed, and system errors must be evaluated. Regression analysis usually yields a calibration curve with confidence limits or tolerance levels.

Data quality, completeness and decisions

Check all instruments to see that they are functioning properly before collecting data for analysis. An equipment malfunction can cause loss of data or inaccuracies. Often some data are missing, or evidence suggests that the data should be adjusted. A vital step after monitoring is to check the reasonableness of the results; assumptions in design, equipment malfunction or misuse, or errors in the analysis can cause results to be unreliable. This step should not wait until after all of the data have been collected. Rather, it must be done early.

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8.1 Production and beneficial use of methane

Biogas is produced during the decomposition of organic matter under anaerobic conditions (see chapter 2.3 'Anaerobic, aerobic and facultative ponds'). It typically contains 60% to 65% methane (CH_4), 35% to 40% carbon dioxide (CO_2) and variable amounts of impurities such as hydrogen sulphide (H_2S) and ammonia (NH_3). Although methane can be flared to reduce greenhouse gas (GHG) impacts (see chapter 8.2 'Greenhouse gas emissions'), it can also be put to beneficial use, offsetting at least some of the farm's electrical and heat energy requirements.

Before you can develop methane capture and use projects, you need to answer the following questions:

- What type of anaerobic digester suits the operation?
- How much biogas will it yield?
- How do the costs and benefits compare with conventional alternatives?

Types of digesters

Large-scale anaerobic digesters in use on dairy farms in the USA and Europe fall into four types:

- Covered anaerobic ponds—traditionally more heavily loaded than conventional anaerobic ponds, with volatile solids (VS) loading rates of up to 170 g m^{-3} and a hydraulic retention time (HRT) of 35 to 60 days (USDA-NRCS 2003). Ponds operate at ambient conditions, so gas yield is reduced in cool seasons (methane production is severely limited in cold climates). Variations incorporating sludge recycling or distributed inflow are referred to as enhanced covered anaerobic ponds.
- Fixed-film digester—a digester, usually heated, containing media that increase the surface area available for bacteria to adhere to, thus preventing washout. As more than 90% of the bacteria are attached to the media, an HRT of days, rather than weeks, is possible. Separation of fixed solids by settling and screening is necessary to prevent fouling.
- Complete-mix digester—sometimes referred to as a continuously stirred tank reactor; usually a circular tank with mixing to prevent solids settling and to maintain contact between bacteria and organic matter. Mixing also maintains a uniform distribution of supplied heat.
- Plug flow digester—a long concrete tank where manure with as-excreted consistency is loaded at one end and flows in a plug to the other end. The digester is heated. Although it can have locally mixed zones, it is not mixed longitudinally.

The total solids content of the effluent stream largely determines the choice between systems. Figure 1 indicates that covered anaerobic ponds and fixed-film digesters suit effluent with up to 3% TS, complete mix-digesters from 3% to 11% TS, and plug flow digesters from 11% to 13% TS. (The term 'digester' is used loosely here to refer to both covered ponds and other types of digesters.)

A number of researchers and commercial developers are currently working on options for recovering energy from manure solids (TS > 20%) using processes such as batch anaerobic digestion, gasification and pyrolysis (GHD 2007b). However, semi-solid material (TS 15%–20% depending on the material) is more problematic as it is too dry to be pumped and agitated but not dry enough to prevent sedimentation and separation of solids and liquids.

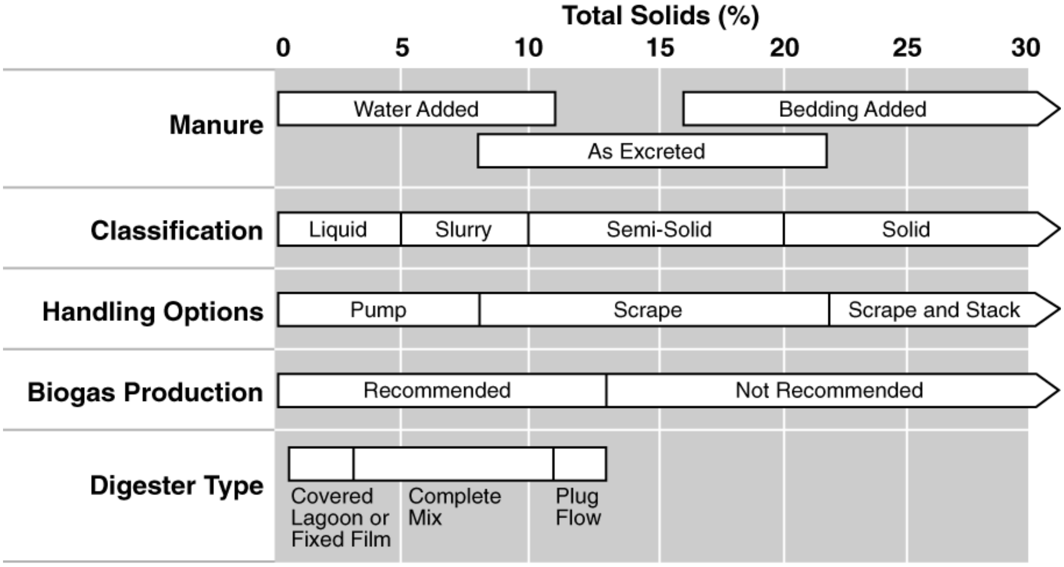

Figure 1. Digester options determined by solids content (source US EPA nd**).**

Figure 2. Enhanced covered anaerobic pond (photo courtesy of George Western Foods).

With a typical solids content of <2% TS, the effluent from most Australian dairies will not be suitable for digestion in complete-mix or plug flow digesters without concentration in a solid–liquid separator (see chapter 2.1 ‘Solid–liquid separation systems’). Most mechanical separators leave most of the volatile solids in the liquid fraction. Improved performance via chemically enhanced separation (see chapter 5 ‘Odour emissions and control’) is an option, but a thorough cost–benefit analysis is necessary.

Biogas yield

In absolute terms, the amount of biogas produced can be calculated from stoichiometrics given that 0.3495 m^3 of methane is produced for each 1 kg of COD destroyed at standard conditions of 0 °C and 101.3 kPa (ASERTTI 2007) or, converted using the Ideal Gas Law, 0.375 m^3 at 20 °C and 101.3 kPa.

Note that although 'methane yield per amount of VS added to the digester' (L CH_4 [kg VS]$^{-1}$) appears to be the most commonly reported measure of methane productivity, it varies with the chemical composition of the VS added. That is, the relative concentrations of carbohydrates, proteins and lipids composing the VS determine the methane productivity. For example, the theoretical methane yield is 415 L CH_4 [kg VS]$^{-1}$ from a carbohydrate but 1014 L CH_4 [kg VS]$^{-1}$ from a lipid (Moller *et al.* 2004). As dairy cattle are fed diets with a higher proportion of poorly digestible lignin and cellulose than pigs, it follows that the methane potential of dairy manure is lower than that from pig manure. In addition, rumen activity results in digestion of the easily degraded materials before excretion. Methane productivity based on the amount of COD added should be therefore be used where possible, or in conjunction with VS (ASERTTI 2007).

In a covered anaerobic pond, the amount of undigested COD (or VS) settling as sludge cannot easily be determined, so $COD_{destroyed} \neq COD_{influent} - COD_{effluent}$. But collection and measurement of methane yield allows the VS degradability to be determined; this information can be used in planning other similar digesters.

Impact of temperature

Although methane can be produced over a wide range of temperatures, microorganisms grow best over a narrower range, so most digesters are designed to operate in the mesophilic temperature range (20–50 °C). Unheated or ambient-temperature digesters in Australia usually operate within the psychrophilic temperature range (10–30 °C). Unfortunately, the ultimate gas yield of psychrophilic digestion (of cattle manure) is, on average, 30% lower than that of mesophilic digestion (Burton and Turner 2003).

In uncovered lagoons, the average water temperature in the upper layer (0–2 m) follows average monthly air temperature, with a slight time lag, but lower layers show a reduced thermal cycle centred on the mean annual air temperature (Hamilton and Cumba 2000). Smith and Franco (1985) describe a model for predicting pond temperatures.

In general, the rate of anaerobic degradation increases at higher temperatures; these reactions follow the Arrhenius temperature-dependence equation:

$$k = A \cdot e^{-Ea/RT} \tag{1}$$

where k = reaction rate constant

A = proportional factor

E_a = activation energy

T = temperature (K)

R = gas constant.

The E_a of most biological reactions is approximately the same (within an order of magnitude); that is, as a simple rule of thumb, the reaction rate doubles with each 10 °C of temperature increase. Therefore, anaerobic degradation at 25 °C should proceed half as fast as at 35 °C. In general, therefore, at lower operating temperatures the hydraulic retention time should be increased in line with the equation above: thus, at a comparable loading rate, the 25-°C digester should have twice the hydraulic retention time of the 35-°C digester.

In reality this applies only to small temperature differences, as other parameters may intensify or diminish the effect. For example, at lower temperatures ammonia is less inhibitory, allowing a faster reaction rate (GHD 2007b). Anecdotal evidence suggests that increased mixing reduces some of the loss in performance at lower temperatures. However, anaerobic degradation is a very complex process and requires complex models to predict digester behaviour accurately.

The lower temperature limit for the methanogens is generally thought to be around 13 to 15 °C, but low activity has been observed at 7 to 9 °C (Shilton 2005). The proportion of methane in the biogas may be low at such low temperatures.

Methane concentration

The theoretical concentration of methane in biogas is:

$$\%CH_4 = 19 \times COD/TOC \quad (2)$$

At typical COD:TOC ratios of 3 to 3.5 in dairy effluent, biogas typically contains 60% to 65% methane (up to 85%). The biogas is usually saturated, and the higher the temperature, the greater is the absolute amount of water vapour held.

Reported yields

The methane productivity of dairy effluent is summarised in Table 1.

Table 1. Reported methane productivity by psychrophilic digestion of dairy effluent.

Methane productivity ($L\ CH_4\ [kg\ VS\ added]^{-1}$)	Details	Source
390[a]	2-ML lagoon, av. 18–19 °C, HRT 67 days, 1.1% TS, loading rate 0.12 kg VS $m^{-3}\cdot day^{-1}$	Safley and Westerman (1992)
194	Laboratory digester, 15 °C, HRT 170 days, loading rate 0.1 kg VS $m^{-3}\cdot day^{-1}$	Safley and Westerman (1994)
103 (128 with recycling of digester contents)	Laboratory digester (fixed-film), 23–24 °C, HRT 2.3 days, 1.3% TS	Powers *et al*. (1997)
70	Laboratory digester (control), 10 °C, HRT 33 days, 0.4% VS, loading rate 0.12 kg VS $m^{-3}\cdot day^{-1}$	Vartak *et al*. (1997)
210[b]	4.6-ML lagoon, av. annual temp. and HRT not specified, loading rate 0.05 kg VS $m^{-3}\cdot day^{-1}$	Craggs *et al*. (2008)
($L\ CH_4\ [kg\ COD\ added]^{-1}$)		
45	14-ML lagoon, 15 °C, HRT 40 days, 0.5% TS, loading rate 0.07 kg VS $m^{-3}\cdot day^{-1}$; only 90% of pond covered	Williams and Gould-Wells (2004)

a: The ultimate methane productivity was 530 L CH_4 [kg VS destroyed]$^{-1}$—much higher than typical.
b: VS added was estimated, not measured.

Where productivity is measured in terms of the amounts of VS added and destroyed, the ratio of the two represents the biodegradability of the VS in the effluent. At a COD:VS ratio of 1.1 in dairy manure (ASAE 1999), the ultimate methane productivity should be approximately 390 L CH_4 [kg VS destroyed]$^{-1}$ (at standard conditions of 0 °C and 101.3 kPa). Although Barth and Kroes (1985) suggest that anaerobic ponds achieve degradation (not removal) of 55% of VS added in dairy effluent, the methane productivity implied by that would be 210 L CH_4 [kg VS added]$^{-1}$, which is at the high end of the reported range. This suggests that VS destruction in ponds is probably lower than 50%.

Beneficial use of biogas

Biogas can simply be flared so that instead of methane (with a GHG equivalence 21 times that of CO_2), the combustion products—CO_2 and H_2O—are discharged. Wotton et al. (2007) reported on the types of flares suitable for biogas and their requirements under Australian regulations.

Alternatively, the biogas could be discharged through a biofilter, where some of the methane will be oxidised by aerobic bacteria (see chapters 5 ‘Odour emissions and control’ and 8.2 ‘Greenhouse gas emissions’).

Unless odour control is a specific aim, it is unlikely that either option will be adopted, as the costs (pond cover, gas collection and flare or biofilter) are not offset by a use with monetary return. In the event of monetary incentives or carbon credits aimed at reducing GHG emissions from manure management systems, the biofilter option may not qualify owing to variable performance and difficulties in measuring reductions.

Hot water

Boilers developed for the combustion of biogas are commercially available. Existing natural-gas-fired boilers can be modified to run on biogas with the following provisions:

- Commercial gas boilers are certified to meet Australian Standards, so modifications will require approval from the Australian Gas Association.
- Boilers containing a copper heat exchanger or fittings will suffer from corrosion unless the biogas is scrubbed to reduce H_2S concentrations. (H_2S in water forms an acid that corrodes metal, especially copper and bronze. Frequent starting and stopping of the boiler intensify this problem.)

Boiler efficiency is typically 80% to 90% (Van Haren and Flemming 2005).

Milk cooling with absorption chillers

Absorption chillers are heat-driven refrigerators relying on heat for energy supply rather than electricity. They are not new technology but have typically been used to supply much larger cooling capacities than are needed for on-farm milk cooling. More recently, the use of absorption chillers for air conditioning has led to the development and commercialisation of smaller units. However, the small, low-temperature units required for on-farm milk cooling to <4 °C are still at the pre-commercial stage and require further development.

Generation of electricity

Systems that generate electricity from biogas consist of:

- an internal combustion engine (compression or spark ignition) or a gas micro-turbine
- an optional heat recovery system
- a generator
- a control system.

Compression (converted diesel) internal combustion engine—Compression engines are also known as dual-fuel engines, as a small amount of diesel (10%–20% of the amount needed for diesel operation alone) is mixed with the biogas before combustion. Dual-fuel engines offer an advantage during start-up and downtime as they can run on anywhere from 0% to 85% biogas (Van Haren and Flemming 2005).

Spark-ignition internal combustion engine—Natural gas or propane engines are easily converted to burn biogas by modifying the carburetion and ignition systems. With

60% methane, biogas reduces engine power by approximately 20%, compared with 10% by natural gas and 5% by LPG (Van Haren and Flemming 2005).

Converted petrol engines offer efficiencies of 18% to 28%, but gas engines and compression (converted diesel) engines offer efficiencies at up to 42%. In general, the small spark-ignition engines (converted petrol engines) cover the range from 10 to 60 kW, dual fuel engines from 40 to 200 kW, and gas engines from 150 to 500 kW (up to 3 MW is possible). An additional 40% of the biogas energy can be captured from engine jacket water and exhaust gases by a heat recovery system (see below).

Gas micro-turbines—These are essentially internal combustion engines with a rotary action instead of reciprocating. Although gas turbines are typically much larger than needed for biogas (e.g. >800 kW), suitably sized micro-turbines have been developed but are currently difficult to source and service in Australia.

A 30-kW (nominal) micro-turbine running on biogas from a covered anaerobic lagoon at California Polytechnic State University (400 cows intensively housed) powered a generator producing 15 to 25 kW at 20% to 25% efficiency (Williams and Gould-Wells 2004). NO_x emissions were 3 ppm (low NO_x emissions are an advantage of micro-turbines over internal combustion engines).

Heat recovery systems—Commercially available heat exchangers can recover heat from the engine water cooling system and exhaust. Typically, heat exchangers will recover around 0.8 kWh of heat per kWh of electrical output from the engine jacket and 0.75 kWh from the exhaust, increasing total (electrical plus thermal) energy efficiency to 45% to 65% (up to 80% in larger installations).

Generators fall into two types: induction (or asynchronous) and synchronous. An induction generator operates in parallel with the mains supply, deriving phase, frequency and voltage from it, and cannot stand alone. Synchronous generators can operate in parallel with the mains or, in the event of supply interruption, without it. Synchronous parallel generation requires a sophisticated interconnection to match generator output to mains phase, frequency and voltage. This is typically more expensive than controls for an induction generation and will attract more scrutiny from the electricity supplier.

A generator may operate without exporting electricity to the distribution grid. Electrical interlocks are used to prevent export and avoid the need for supplier approval (local electrical contractors are capable of installing interlocks, and commonly do so for backup generators).

Common problems with using biogas

Water vapour can interfere with pressure reducers, boiler orifices and other devices, and reduce the energy value of the biogas. Condensate traps offer an effective way of removing moisture (Van Haren and Flemming 2005).

H_2S is corrosive even in small concentrations. To avoid corrosion, H_2S levels should not exceed 500 ppm for use in conventional internal combustion engines (Van Haren and Flemming 2005), although manufacturers may accept up to 1000 ppm (Harding and Olliff 2007). However, reported H_2S concentrations in biogas from dairy digesters have been as high as 6000 ppm, so H_2S removal may be required. The cost of H_2S removal is up to 20% of generation plant cost (Harding and Olliff 2007), but must be considered in conjunction with the proposed maintenance schedule.

Biogas is not easily compressed. Therefore, it is difficult to use the biogas for anything but (nearly) continuous on-site consumption.

Back-up gas supply

Contingency plans must be developed for systems reliant on continuous biogas supply. Generators isolated from the grid, biogas-fired boilers for hot water and, in the future, absorption chillers for milk cooling will require alternative gas sources when the system is off-line for maintenance. It is inadvisable to select equipment reliant on biogas where winter yields are unable to meet expected energy needs.

Energy budgets for large- to medium-sized dairies

For medium- to large-scale dairies where cows are not housed intensively, the energy content of the collected manure may not be sufficient to justify the costs incurred to generate electricity. These farms may prefer to focus on offsetting the energy requirements for hot water and, if suitable absorption chillers become available, milk cooling. The energy use of these two activities accounts for three-quarters of a dairy’s energy bill: 43% for milk cooling and 33% for hot water (Rogers and Alexander 2000).

To show the feasibility of offsetting energy requirements, Table 2 compares energy yield following methane capture with typical requirements for hot water alone, or for hot water and milk cooling, under the following assumptions:

- 4.7 kg VS per cow day^{-1} (derived from Nennich *et al.* (2005) in chapter 1.2 ‘Characteristics of effluent and manure’ and reduced by a safety factor of 20%)
- 20% of VS removed by pre-treatment
- boiler efficiency 80%, chiller COP 0.6
- hot water use of 3 L per cow day^{-1} (input at 15 °C, output at 90 °C)
- milk yield of 20 L per cow day^{-1} cooled from 17 °C (after platecooler) to 4 °C
- sufficient gas storage available to supply gas to appliances whenever needed (i.e. no loss of gas to a flare).

Table 2. Percentage of daily manure output that must be collected to satisfy use.

Use	Assumed average annual methane productivity (L CH_4 [kg VS added]$^{-1}$)						
	50	**75**	**100**	**125**	**150**	**175**	**200**
Hot water only	18	12	9	7	6	5	5
Hot water + milk cooling	40	27	20	16	14	12	10

As the energy requirements are modelled on a per-head basis, the results in Table 2 are independent of herd size. However, system cost and payback period will vary with herd size, and economies of scale are expected.

Methane productivity is a critical determinant of whether sufficient biogas is available to meet energy requirements for hot water and milk cooling. Although the yields in Table 1 suggest that Australian covered ponds operate somewhere within the range assumed in Table 2, further research is required to more accurately determine the methane productivity expected over the range of climatic conditions in each dairy region.

As the values of methane productivity in Table 2 are annual averages, actual productivity will vary significantly from summer to winter. As mentioned above (see section ‘Impact of temperature’), a 15-°C drop in monthly average temperature from summer to winter could more than halve the methane productivity. Winter operation must be investigated to ensure that sufficient energy is available to supply biogas-reliant equipment. Further research is required to identify this productivity–temperature dependence in order to avoid overuse of the back-up gas supply.

After sufficient biogas is generated to meet the requirements for hot water (and milk cooling), additional biogas is likely to be flared. Alternatively, it can be burned in the

boiler, and a heat exchanger can be used to transfer the heat to the covered pond, although it may have a negligible impact on pond temperature.

Considerations for the overall effluent management system

Solids separation before digester

Digester function will be compromised by indigestible solids, so the use of a shallow trafficable solids trap or sand trap to remove large material is wise. Sand-bedded freestalls will require significant investment in solids separation equipment (see chapter 2.1 ‘Solid–liquid separation systems’). However, the removal of volatile solids along with the fixed solids will reduce the potential methane yield, so a trade-off is necessary.

Waste feed and organic bedding may be (slowly) biodegradable, but long straw and floatable material should be held back from the digester. Rundown screens with a large orifice size (see chapter 2.1 ‘Solid–liquid separation systems’) may be useful in this regard, as typical VS removal is low.

The effluent stream must be free of extraneous objects that could cause blockages, as a covered pond is not accessible without significant disruption and downtime.

Desludging and crust management

Once a pond is covered, there is no easy way to gain access for maintenance activities such as desludging. Some manufacturers include a ‘zippered’ opening at intervals around the pond perimeter, into which a sludge agitator and pump can be inserted when necessary. This is, at best, only a back-up option, and the design of the digester and its crust- or sludge-handling systems must be robust and reliable.

It is not economically feasible to cover a pond with enough ‘dead’ space for sludge accumulation to match the design life of the cover material. Indeed, there is significant economic advantage in minimising sludge accumulation as much as possible. Regular sludge harvesting or removal via a network of pipes across the pond base is an option in some enhanced covered anaerobic ponds and covered in-ground anaerobic reactors (see chapter 2.8 ‘Desludging and pond closure’). Anecdotal evidence suggests that the recirculation of the collected sludge increases biogas production (by improving contact between bacteria and substrate).

Unfortunately, much of the design information relating to sludge and crust management is proprietary and tightly held by digester companies.

Storage

Effluent storage and reuse are still required, as pollutant concentrations after digestion can exceed the standards for discharge.

Safety precautions

Methane is odourless and colourless and is explosive when mixed with air at 5% to 15% by volume. Be aware that whereas methane is lighter than air and will disperse, CO_2 and H_2S are heavier than air and can collect in confined spaces (see chapter 6 ‘Occupational health and safety’). Biogas equipment areas should be open or well ventilated to disperse fugitive gases. Safety precautions must be considered during design and maintenance. Procedures and equipment for a ‘hazardous area’ classification might have to comply with AS 2430.3.7 (Standards Australia 2004). Specific safety procedures are beyond the scope of this document; seek specialist advice.

Design criteria and selecting a designer

USDA-NRCS (2003) provides basic design criteria (HRT and VS loading rate) for locations across the USA, but no comparable guidelines exist for Australia. Further research is needed to develop similar tools for local conditions.

Specialist knowledge and experience in the design of anaerobic digesters are essential for a successful outcome. Early adopters (before 1982) in the USA experienced a 75% failure rate in plug flow and complete-mix digesters and a 30% failure rate in covered ponds (US Department of Energy 1995). Inadequate design was cited as one of the main reasons for failure. Other reasons varied but included shutdown due to declining energy prices and sale of the farm.

Although the skills and experience needed to develop a biogas project are slowly becoming more accessible, the technological and financial risks resulting from poor advice are still significant. Refer to published information including (but not limited to) US EPA (nd.) and ASERTTI (2007) when comparing proposals.

Current research

Given the concerns regarding climate change (see chapter 8.2 'Greenhouse gas emissions'), anaerobic digesters are attracting significant research and commercial interest. Magma (2007) reviewed R&D activities for the Rural Industries Research and Development Corporation's 'Methane to Markets in Australian Agriculture' program. The review includes details of previous studies of the feasibility of installing covers on piggery anaerobic ponds, flares, scrubbers, porous burners and fuel cells.

Additional information is provided in a report by GHD (2007a) that reviews options for methane capture and use in Australian intensive livestock industries. The report considers the viable project scale for dairy farms, using European and US data from Mehta (2002), who assumed digester and engine costs of $\$6250 \cdot kW^{-1}$ and 0.15 kW per animal or $1000 per animal. Although the results suggest a payback period of 6 years, the 'power generation potential' seems to be based on heated (mesophilic) digesters. Covered anaerobic ponds (the most appropriate approach for dilute effluents in Australia) are typically psychrophilic and will have much lower gas yields. Predictions by RCM Digesters (2003) for a covered pond (see 'Case studies' below) suggest a power generation potential of 0.05 kW per animal from an ambient-temperature pond in northern Victoria.

GHD (2007a) concluded that the economic viability of methane projects is highly variable and requires site-specific analysis. In general, sites that currently use gas (natural or LPG), or could, will more likely be feasible and require less investment. Electrical generation may not be feasible for smaller sites.

Case studies

1—2200-cow freestall northern Victoria

(RCM Digesters Inc. 2003) studied four options to produce and use methane at a proposed 2200-cow freestall dairy in northern Victoria:

- Option 1—Covered anaerobic pond, freestalls bedded with sand, floodwashed
- Option 2—Option 1 with organic matter bedding replacing sand
- Option 3—Option 2 plus cheese whey from local factory
- Option 4—Plug flow digester using scraped manure.

Although the initial motivation to investigate biogas production was odour control, the proponent liked the opportunity to offset energy consumption. The financial analysis of

the four options showed negative net present values (NPV) with low internal rates of return (IRR) for Options 1, 2 and 3, but Option 4 was profitable. The proponent chose not to adopt Option 4 owing to an aversion to scraped manure systems. Of the flushed manure options, Option 3 returned the best NPV at –$58 860.

Table 3. Analysis of digester options (RCM Digesters Inc. 2003).

	Option 1	Option 2	Option 3	Option 4
Av. gas production ($m^3 \cdot day^{-1}$)	695	1 580	1 711	3 388
Av. electrical output (kWh)	51	116	126	214
Electrical offsets (% of bill)	36	81	88	154
Max. heat recovery (MJ)	320	910	1 095	807
Min. heat recovery (MJ)	248	416	502	179
Capital cost ($)	626 609	796 569	797 193	1 201 228
NPV ($)	–280 860	–105 466	–58 860	391 973
IRR (%)	0.0	2.7	6.0	42
Simple payback period (y)	20	11	10	6.5

The analysis used the following assumptions:

- Costs of anaerobic pond earthworks were not included in the capital cost.
- $61 870 was subtracted from the capital cost as a result of avoiding a permeable cover required for odour control.
- Electrical consumption offset was valued at $\$0.102 \cdot kWh^{-1}$; excess was sold at $\$0.075 \cdot kWh^{-1}$. Both values include renewable energy certificates of $\$0.04 \cdot kWh^{-1}$.
- Operation and maintenance costs were $\$0.017 \cdot kWh^{-1}$; energy costs increased at 3% p.a.
- 100% finance, 10-year loan period, 8% loan interest rate.
- Discount rate of 11%.
- 15-year project life; system downtime 10%.
- Exchange rate of A$0.658 per USD.
- The financial benefit resulting from the reuse of treated water was not included. For Option 4, the sale of digested solids was estimated at $49 924 and included in the financial analysis.

The proposed loading rate for the anaerobic pond was low (even after 10 years' sludge accumulation), and the resulting large surface area would incur unnecessarily high cover costs.

The predicted gas yield exhibited a seasonal trend. Electrical output met anticipated consumption during summer but fell to around two-thirds of consumption over winter. The project has been given planning approval but construction has been delayed.

2—US feasibility study

Although the nature of the Australian dairy industry is different from that of the USA, a report by the (US Department of Energy 1995) provides an insight into the effect of scale and the proportion of manure collected on economic viability. The report analysed the feasibility of methane recovery at three different sized farms, each collecting either 15% or 55% of the manure generated. Table 4 shows the results for covered lagoons.

Table 4. Impact of scale on economic feasibility of methane recovery (US Department of Energy 1995).

Herd size (head)	Proportion collected (%)	NPV ($US)	IRR (%)	SPP (y)
250	15	–16 545	1.1	13.3
500	15	–7 744	6.0	8.5
1000	15	11 253	10.8	6.4
250	55	–7 912	6.2	8.3
500	55	12 507	10.8	6.4
1000	55	49 891	13.9	5.6

The analysis used the following assumptions:

- Revenue was based only on savings from offset electrical and heating use, and surplus electricity sales (if available).
- Biogas yields were not specified but were calculated for central Texas.
- 15-year project life.
- O&M costs increased at 1.5% p.a.; energy cost did not increase.

The study concluded that the minimum herd size required to achieve an IRR of 8.5% was 780 to 890 head if 15% of manure is collected, decreasing to 400 to 560 head if 55% is collected.

3—Mobile fixed-film digester

Active Research (Active Research 2007) used effluent from the Victorian DPI's Ellinbank farm to investigate methane production from a mobile fixed-film digester. The 2220-L digester operated at 38 °C, had a hydraulic residence time of between 15 and 100 h, and used ultrasound to 'disintegrate floc'.

The 1-kW, 24-kHz ultrasound unit resulted in a 30% increase in gas production. Unfortunately, incomplete data prevented verification of this result and the methane productivity results.

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8.2 Greenhouse gas emissions

With no short-term effects on the local environment and little direct consequence to the farm economy, greenhouse gas (GHG) emissions have received little attention until recently. However, concern over climate change and the likelihood of requirements for GHG mitigation has meant that GHG emissions are now a priority for most sectors of the economy.

Dairy farming's GHG footprint

Agriculture was responsible for 16% of Australia's 559 Mt of GHG emissions in 2005 (AGO 2007b) and is the dominant sector for emissions of methane (CH_4; 60%) and nitrous oxide (N_2O; 85%). Livestock emissions represent 71% of the agricultural sector subtotal, or 11% of national emissions.

Enteric fermentation in the animal gut is responsible for the majority of livestock GHG emissions. According to the Australian Greenhouse Emissions Information System (AGEIS; AGO 2007a), direct emissions from the dairy industry in 2005 included 7266 kt CO_2-equivalent (CO_2-e) from enteric fermentation, 815 kt CO_2-e from N loss from dung and urine voided to soils, 574 kt CO_2-e from manure management, and a further 100 kt CO_2-e from application of effluent and manure to soil. Additional indirect N_2O emissions are attributed to the dairy industry; further information is provided by the National Greenhouse Gas Inventory Committee (2006) and Mosier *et al.* (1998). Further information on emissions from enteric fermentation is provided by Hegarty (2001) and Eckard *et al.* (2002).

CH_4 and N_2O are emitted during the management of effluent and manure. The 574 kt CO_2-e from the dairy industry in 2005 comprised 26.8 kt of CH_4 and 34.3 t of N_2O with a CO_2-e of 563 and 10.6 kt respectively. CH_4 has a global warming potential 21 times that of CO_2, and N_2O 310 times that of CO_2 (EPA Victoria 2002). These emissions represent 17% of all livestock manure management and 0.1% of Australia's total GHG emissions.

Eckard *et al.* (2002) developed a decision support framework for the Australian dairy industry and suggested that emissions from effluent ponds contribute <1% to 2% of farm GHG emissions, mostly as enteric CH_4 and N_2O from soils, fertiliser and urine. The contribution from effluent ponds suggested by Eckard is much less than the 7% suggested by AGEIS (or 5% if indirect emissions are included). Some of the difference may be due to the use of empirical emission factors by the National Greenhouse Gas Inventory Committee (2006); for example, the methane productivity of 0.24 m^3 CH_4 [kg VS added]$^{-1}$ is at the upper end of the range suggested in chapter 8.1 'Production and beneficial use of methane'. Eckard *et al.* (2002) suggested that further research is needed to verify the decision support model under Australian conditions, as the level of uncertainty in modelled emissions may range from 20% to 300%.

Generation of GHGs

GHG emissions must be separated into direct and indirect emissions. Direct emissions arise from sources such as enteric fermentation and manure management. Indirect emissions are caused by the use of electricity. Although this chapter concentrates on minimising direct emissions arising from manure management, it is important to recognise that there are opportunities to minimise indirect emissions through improvements in energy use efficiency in the dairy, and that most environmental authorities now require an energy audit and action plan as part of an application for regulatory approval. Options for improving energy efficiency may include things such as:

- platecoolers and cooling towers to reduce heat load in milk

- variable-speed drives on milk pumps and vacuum pumps
- energy-efficient lighting
- high coefficient of performance (COP) refrigeration systems.

Further information can be found at http://www.cowtime.com.au/EnergyMonitor/index.aspx

Emissions from the on-site use of delivered gas (LPG, propane, natural gas) are also direct and should be considered as part of the energy management system.

AGEIS suggests that within manure management in the dairy industry, CH_4 is a substantially larger contributor to GHG emissions than N_2O. Although both depend on microbial activity, there are fundamental differences in their modes of formation that must be appreciated before control strategies can be considered.

CH_4 is a product of the anaerobic decomposition occurring in most agricultural effluent treatment ponds (see chapter 2.3 'Anaerobic, aerobic and facultative ponds'). Under anaerobic conditions, temperature and storage time determine the amount produced (see chapter 8.1 'Production and beneficial use of methane'). As CH_4 is poorly soluble in water, it is easily lost into the atmosphere.

In contrast, N_2O is not directly produced from compounds primarily present in manure. N_2O is a product of incomplete denitrification: the conversion of nitrate into N_2 under anaerobic conditions. However, raw effluent nitrate concentrations are typically low to negligible (and the nitrification step converting ammonium into nitrate is limited by a lack of oxygen), so anaerobic ponds produce little N_2O. N_2O may also be an intermediary product of nitrification under suboptimal conditions: low oxygen availability, high NH_3 concentrations, low C:N ratios (Monteny *et al.* 2001).

Solid manure stockpiles, however, are more likely to undergo uncontrolled nitrification and denitrification resulting in N_2O emissions. Stockpiles have higher concentrations of nitrate as a result of aerobic activity, but when they are poorly composted or periodically saturated by rain, anaerobic conditions can result in denitrification. Incomplete denitrification and emission of N_2O is favoured by low COD-to-NO_3 ratios, low pH and the presence of oxygen (Shilton 2005).

The National Greenhouse Gas Inventory Committee (2006) has adopted emission factors of 1 g N_2O [kg N]$^{-1}$ for anaerobic ponds and 20 g N_2O [kg N]$^{-1}$ for solid storage. Although the accuracy of the assumptions is debateable, the magnitude of difference illustrates that it is important for the total GHG emissions to be considered rather than CH_4 and N_2O in isolation. That is, removing solids from effluent before the anaerobic pond may reduce its CH_4 emissions, but the impact of any increase in N_2O emissions (with its larger global warming potential) must still be considered. System choice must not simply transfer emissions from one component to another for accounting purposes, but rather consider the entire production system.

Ammonia is not a direct GHG but it does have implications for odour, atmospheric N deposition ('acid rain') and eutrophication. Atmospheric deposition of N has been shown to enhance biogenic N_2O formation in Europe, so it can have an indirect impact on GHG emissions. Mosier *et al.* (1998) assumes that 1% of NH_3 emissions are transformed into N_2O. N_2O also contributes to stratospheric ozone depletion.

Direct emissions of CO_2 from manure are not considered to contribute to global warming, as the carbon released originates from the fixation of atmospheric carbon in plant material and cycles over a relatively short period of time (Pattey *et al.* 2005).

GHG control strategies within manure management

Amon *et al.* (2006) compared different manure management strategies. More than 90% of GHG emissions from managing untreated dairy slurry originated from CH_4 produced during storage (80-day retention). Amon *et al.* (2006) therefore concluded that GHG

abatement measures are most effective if they reduce CH_4 emissions during storage. Covering the storage with straw increased CH_4 and N_2O emissions and resulted in the highest total GHG emissions (Table 1). Treatments involving anaerobic digestion, aeration and solids separation were effective at reducing GHG emissions compared with untreated slurry. GHG emissions from the separated treatment comprised 41.3 kg CO_2-e during storage, 14.8 kg CO_2-e from the composted solids (pile turned seven times in 80 days) and 2.4 kg CO_2-e during reuse.

Table 1. Emissions from variously treated dairy slurry (storage, solids stockpile and reuse).

	NH_3 (g m^{-3})	% of un-treated	CH_4 (g m^{-3})	% of un-treated	N_2O (g m^{-3})	% of un-treated	Total GHG (kg CO_2-e m^{-3})	% of un-treated
Untreated	227	100	4047	100	23.9	100	92.4	100
Separated [a]	403	178	2363	58	28.6	120	58.5	63
Digested	230	101	1345	33	31.2	130	37.9	41
Straw cover	320	141	4926	122	52.5	220	120	130
Aeration	423	186	1739	43	54.2	227	53.3	58

[a] Liquid and solid phases

Similarly, increases in GHG emissions following the application of straw covers to slurry storage were reported by Berg *et al.* (2006) (lab, pig slurry) and Cicek *et al.* (2004) (farm, piggery lagoons). Possible reasons for the increases include sinking straw providing an additional C source for methanogens, and that reduced surface mixing maintains optimum anaerobic conditions, both of which result in increased CH_4 emissions. In addition, straw at the interface between N-containing slurry and the atmosphere provides an environment for uncontrolled nitrification and denitrification and N_2O emission. However, conflicting results exist: Sommer *et al.* (2000) (cattle slurry) and Lague *et al.* (2004) (piggery lagoon) measured reduced CH_4 emissions following the addition of a straw cover to storages.

Given the uncertainty over possible augmentation of GHG emissions via straw covers, it is fortunate that dairy effluent can be naturally self-crusting and offers the potential for bacterial oxidation of methane without requiring additional carbon. Petersen *et al.* (2005) demonstrated that oxidation can remove CH_4 under practical storage conditions, and Petersen and Ambus (2006) determined maximum fluxes of ~1 g CH_4 $m^{-2}\cdot day^{-1}$ from natural crusts on cattle slurry. Although the CH_4 flux from an uncovered storage was not directly compared by Petersen and Ambus, other researchers have found fluxes from an uncovered storage averaging 28 to 31 g CH_4 $m^{-2}\cdot day^{-1}$ (Sneath *et al.* 2006) and a maximum of 18 g CH_4 $m^{-2}\cdot day^{-1}$ (Sommer *et al.* 2000).

Composting of solids separated from the effluent stream before the anaerobic pond appears to offer some reduction in overall GHG emissions, but some research has demonstrated negative results. Separated solids must undergo true aerobic composting to mitigate GHGs. Minimal intervention ‘composting’ or stockpiling solids without turning is a simple and effective, albeit slow, means of reducing volume and volatile solids. However, it is not a uniformly aerobic process and should not be termed composting; anaerobic conditions do exist, leading to the production of CH_4. Lopez-Real and Baptista (1996) found that forced aeration and turned windrows were effective composting procedures and substantially reduced CH_4 emissions compared with static stockpiles. Therefore, if the separated solids are not composted with due attention to C:N, porosity and moisture content, CH_4 emissions would remain high, and additional N_2O emissions might be produced as a result of incomplete denitrification or nitrification under unfavourable conditions. Dinuccio *et al.* (2008) demonstrated such a result, finding that the sum of GHG emissions from the liquid and separated solids fraction (with 21% solids in a static stockpile) was 25% higher than that from an untreated control (pig manure).

However, Pattey *et al.* (2005) found that emissions of CH_4 and N_2O were higher from anaerobic dairy slurry (397 g CO_2-e [kg DM]$^{-1}$) than from composted dairy manure

(static pile fitted with two air supply pipes; mostly aerobic; 207 g CO_2-e [kg DM]$^{-1}$) over a 90-day storage period. Emissions from stockpiled dairy manure solids (partially anaerobic, partially aerobic) fell between the other two treatments at 301 g CO_2-e [kg DM]$^{-1}$. The contribution of CH_4 decreased and N_2O increased with increasing oxygen availability. Unfortunately, little information on the manure was provided.

Hao *et al.* (2004) measured GHG emissions from composted beef feedlot manure (with straw bedding). Although emissions (197.5 g CO_2-e [kg DM]$^{-1}$) were similar to those reported by Pattey *et al.* (2005), the relative contribution of CH_4 compared to N_2O was significantly different (17% due to CH_4 in Pattey *et al.* (2005), 95% in Hao *et al.* (2004)). Although slightly different global warming potentials were used, the reason for the magnitude of the difference is unclear.

Developments in the use of deep-litter dairy housing in intensive operations in colder climates may avoid some CH_4 emissions as a result of not requiring large anaerobic ponds, but may result in increased N_2O emissions from the decomposition of solids under uncontrolled, predominantly aerated conditions (Monteny *et al.* 2006). Insufficient evidence is available to draw any conclusions for Australian conditions.

Summary of management options for manure treatment and storage

The following two strategies are the most effective means of reducing GHG emissions from manure:

- **Manure minimisation.** Minimise the volume of manure produced by 'ensuring that the energy requirements of the animals are met from the highest digestibility feed available, fed only at levels required for the desired animal performance' (Hegarty 2001). The less the amount of VS and N to be decomposed, the less will be emissions of CH_4 and N_2O.
- **Impermeable cover on anaerobic ponds with biogas combustion.** Collection and combustion of CH_4 produced in anaerobic ponds reliably offers the most effective reductions in GHG emissions, as the digester is essentially a closed vessel and only the products of combustion are emitted to the atmosphere. Burning one molecule of CH_4 yields one molecule of CO_2, reducing the global warming potential by a factor of 21. Biogas capture enables the generation of heat or electrical energy with direct economic benefits (see chapter 8.1 'Production and beneficial use of methane').

Where biogas capture is not an option, the following options are effective (albeit to a lesser degree):

- **Direct application, or minimising retention time.** Direct application avoids the anaerobic storage of effluent and the production of CH_4. However, aside from the likelihood of increased emissions of N_2O from the reuse area, the possible pollution of surface and groundwaters creates more risk than it resolves. There is, however, merit in reducing retention time whenever possible by distributing effluent regularly outside of the storage period.
- **Properly managed composting.** Maintain process parameters within recommended ranges to that ensure aerobic conditions prevail (see chapter 2.9 'Composting'). Stockpiling solids without frequent turning is not composting.
- **Separation of solids.** If separated solids are composted, solid–liquid separation may reduce overall GHG emissions.
- **Retain crusts on anaerobic ponds.** Crusts provide an environment for bacterial oxidation of methane.

- **Permeable covers.** Gas-permeable covers have shown mixed results, but where a crust cannot be maintained, they may warrant further research (see chapter 5 'Odour emissions and control').

GHG control strategies within effluent and manure reuse

Saggar *et al.* (2004) summarise research on N_2O emissions from pasture soils following the application of animal effluent and manures. Emissions vary greatly depending on water-filled pore space and climatic conditions. Reported emissions ranged from <0.1% to over 10% of effluent N applied.

The nature of the effluent applied is also important. Bhandral *et al.* (2007) found that untreated dairy effluent lost a lower percentage of applied N as N_2O than treated effluent (0.7% vs. 2%, autumn application), even though three times as much N was applied in the former (61 vs. 21.8 kg N ha^{-1}). Untreated effluent has a larger proportion of organic N, which will decompose (mineralise) gradually to inorganic N.

Saggar *et al.* (2004) suggest that reuse management practices are likely to affect individual gases differently. For example, direct injection of sludge may reduce NH_3 emissions but increase losses of N_2O depending on C and N levels and moisture status. Although this area is currently the subject of considerable research, strategies that increase the efficiency of N use generally are appropriate for minimising N_2O emissions upon reuse.

Further information on minimising N_2O emissions from animal agriculture is presented by de Klein and Eckard (2008). Best management practices for soils are listed at http://www.nitrogen.unimelb.edu.au/index.htm.

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8.2 Greenhouse gas emissions

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Appendix A

Reference	Separation efficiency (%) [g]							TS in influent (%)	TS in solids (%)	Flow (L/min)	Notes
	TS	TVS	TKN	Org N	NH_4	TP	K				
Gravity - settling in laboratory											
Chastain *et al.* (2005)	47 **62** 35 **42**	55 **67** 42 **48**	21 **43** 13 **22**	63 **73** 39 **46**	-4[a] **25** [a] -6 [a] **5** [a]	26 **47** 43 **49**	-5 **24** -5 **6**	1.7 0.7	3.8 2.9	batch batch	"Milking center washwater" only was sampled from a commercial US pasture based dairy for laboratory settling, 30 minute retention. Results are % in bottom 28 and 11% of volume respectively.
Converse and Karthikeyan (2004)	53		34		26	43	22	1.3	not reported	batch	Flood wash from freestall, sampled for lab study using 13 L settling columns. Data shows % in bottom 25% of volume, 60 minute retention.
Chastain *et al.* (2001)	55	55	24	26	21	28	1	4.2	80	batch	Flood wash from a freestall dairy (organic bedding) settled for 30 minutes in 1L Imhoff cones.
Barrow *et al.* (1997)	**63**		**22**			**60**	**41**	1.0	not reported	batch	Excreted manure/urine mix diluted to 1.0% TS and settled for 20 mins 1L in Imhoff cones.
Gravity - sedimentation basin											
Meyer et al. (2004)	59							1.5	36 - 14, ave. 19[b]	5720	Flood wash from a freestall dairy (with composted manure bedding) collected in a settling basin (2 bays x 134 x 16.5m) with 'weeping walls' for side walls.
Barrow *et al.* (1997)		48	21			17	2	0.6	not reported	not reported	Excreted manure/urine mix diluted to 1.0% TS, paired sedimentation basins (details not given).
Burcham *et al.* (1997)	59 [c]		49 [c]			43 [c]		not reported	not reported	not reported	Flood wash from sand bedded freestall, 15 kL sedimentation basin.
Gravity - Geotube											
Worley *et al.* (2005)	**97**		**80**	**92**	**44**	**79**	**36**	5.3	19	batch	Effluent pumped from sludge layer of anaerobic lagoon.

Reference	Separation efficiency (%) [g]							TS in influent (%)	TS in solids (%)	Flow (L/min)	Notes
	TS	TVS	TKN	Org N	NH_4	TP	K				
Mechanical - Inclined stationary screen											
Wright (2005)	**63**							0.7	4.3	11130	Flushed effluent, 1.3 mm screen.
Zhang *et al.* (2003)	27	34						2.2	not reported	1328	Flood wash from a freestall dairy (with composted manure bedding) pumped to a 2mm screen.
Zhang and Westerman (1997)	*49* *68*							4.6 2.8	12 6	not reported	1.7 and 0.6 mm screens respectively, reported by USDA but not referenced.
Mechanical - Elevating stationary screen											
Chastain *et al.* (2001).	61	63	49	52	46	53	51	3.8 [d]	20	not reported	Flood wash (clean water), freestall dairy (organic bedding) to a 1.5mm screen.
Fulhage & Hoehne (1998)	*46*	*50*	*17*	*19*	*8*	*11*	*10*	not reported	23	not reported	Flood wash, 1.5 mm screen.
Barrow *et al.* (1997).	32 58	30	12 43			28 56	10 24	0.6 barn 0.5	not reported	not reported	Flood wash from sand bedded freestall or parlour, 1.5 mm screen.
Burcham *et al.* (1997).	47 [c]		36 [c]			42 [c]		not reported	not reported	not reported	Flood wash from sand bedded freestall, 1.5 mm screen.
Mechanical - Screw press											
Gooch *et al.* (2005) Farm AA (0.5 mm) Farm ML (2.25 mm) Farm FA (0.75 mm)	 **50** **4** **71**	 **56** **5** **77**	 **16** **1** **24**	 **18** **2** **29**	 **14** [e] **1** [e] **20** [e]	 **24** **1** **24**	 **13** **1** **-**	 7.5 5.5 10.0	 23.7 29.3 25.3	 170 212 187	Three screw presses, used on three different farms; farm AA & ML processed digester effluent, farm FA processed raw effluent.
Converse *et al.* (1999)	19 **20** [f] 47 **63**		 **4** **29**			 **3** **28**	 **1** **10**	1.0 10.1	28.3 27.4	622 163	Freestall and tie stall barns with mechanical scrapers, manure diluted to range 1 to 10%. Screw press fitted with 2.4 mm screen.

Reference	Separation efficiency (%) [g]							TS in influent (%)	TS in solids (%)	Flow (L/min	Notes
	TS	TVS	TKN	Org N	NH4	TP	TK				
Combination - Screen + sedimentation basin											
Chastain *et al.* (2001).	70	73	51	61		60	48	3.8	20screen 11 basin	not reported	Flood wash (clean water), freestall dairy (organic bedding) to a 1.5mm screen plus paired basins.
Barrow *et al.* (1997).	55 71		14 45			40 75	10 47	0.6 barn 0.5	not reported	not reported	Flood wash from sand bedded freestall or parlour, 1.5 mm elevating stationary screen, paired sedimentation basins.
Burcham *et al.* (1997).	91 [c]		48 [c]			67 [c]		not reported	not reported	not reported	Flood wash from sand bedded freestall, 1.5 mm paddle conveyor screen plus 15 kL sedimentation basin.
Combination - Screen + sedimentation basin + lagoon											
Chastain *et al.* (2001).	93	96	74	91	54	86	60	3.8[d]	20screen 11 basin 7sludge	not reported	Flood wash from a freestall dairy (with organic bedding) pumped to a 1.5mm screen, basin and lagoon details not provided.

[a] TAN = $NH_4 + NH_3$
[b] Average of 6 samples taken at progressive distance from basin inlet.
[c] Estimated from graphs as tabulated data not presented.
[d] These figures are unusually high (46% removal of ammonium and 51% removal of K) as a large amount of bedding (12 times normal amount) and waste feed was present in the flushed manure giving an unusually high 3.8% TS. The authors suggest this readily removable material may have assisted with the removal of soluble nutrients by absorption and additional filtration.
[e] Reported as NH_3.
[f] Discrepancy in paper calculation.
[g] Separation efficiencies shown in italics did not specify method of determination, those calculated by mass balance are shown in bold, otherwise determined by approximate/concentration reduction method.

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Appendix B

Planning Requirements for a Feedpad

- -- Local Government Planning Approval
- – buffer distances to neighbouring properties and residences
- – buffer distances to watercourses
- – buffer distances to roads
- – noise
- – odour
- – lighting
- – road access and slight distances
- – stormwater control
- – effluent management
- – building standards
- – hours of operation
- – aesthetics
- – vegetation management/clearance of trees.

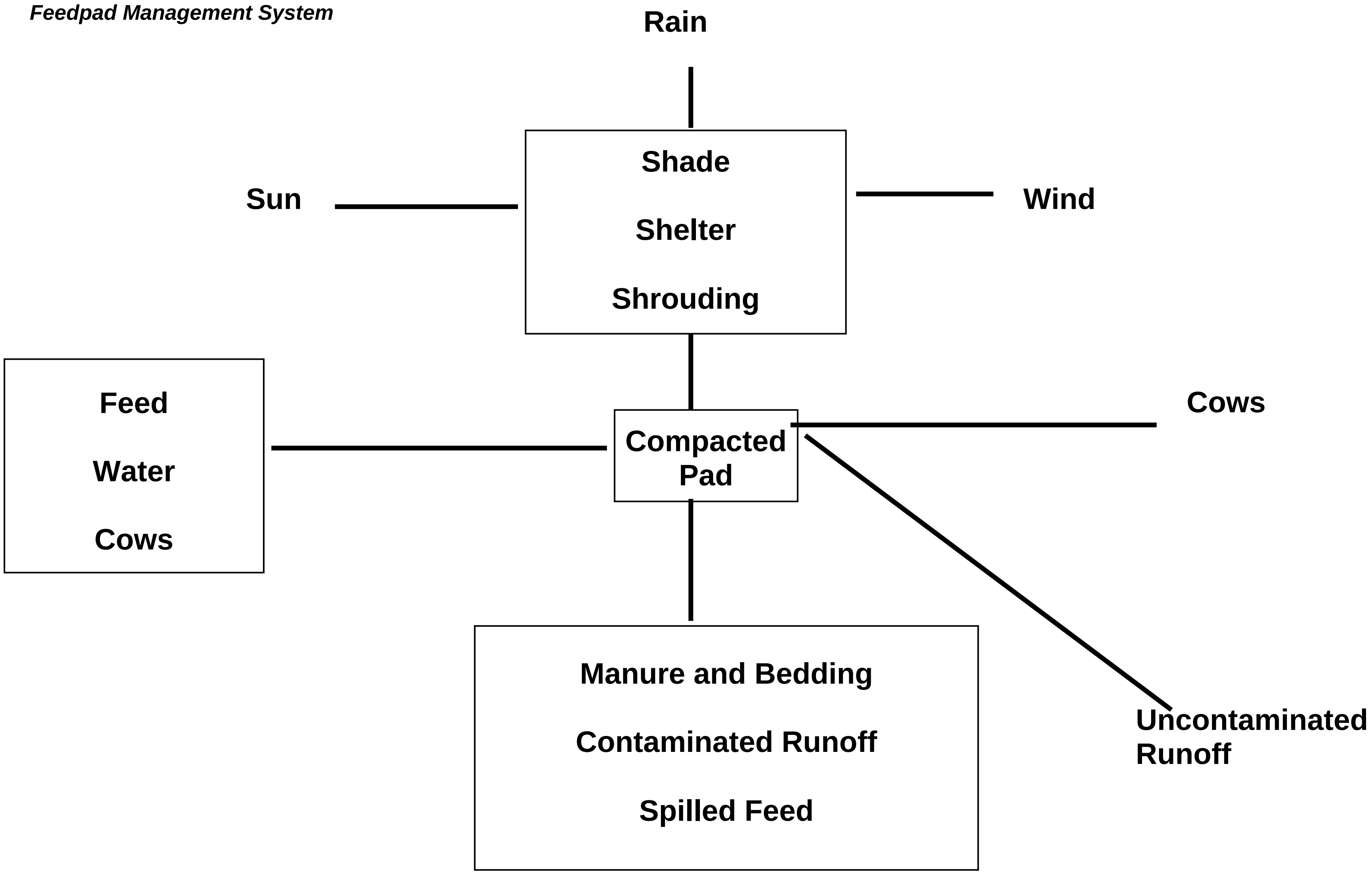
Dairy Company Quality Assurance

- -- 3Milk Harvesting Regulations
- – distances to the milk room from the facility
- – distances to effluent management system
- – quality control for milk harvesting.

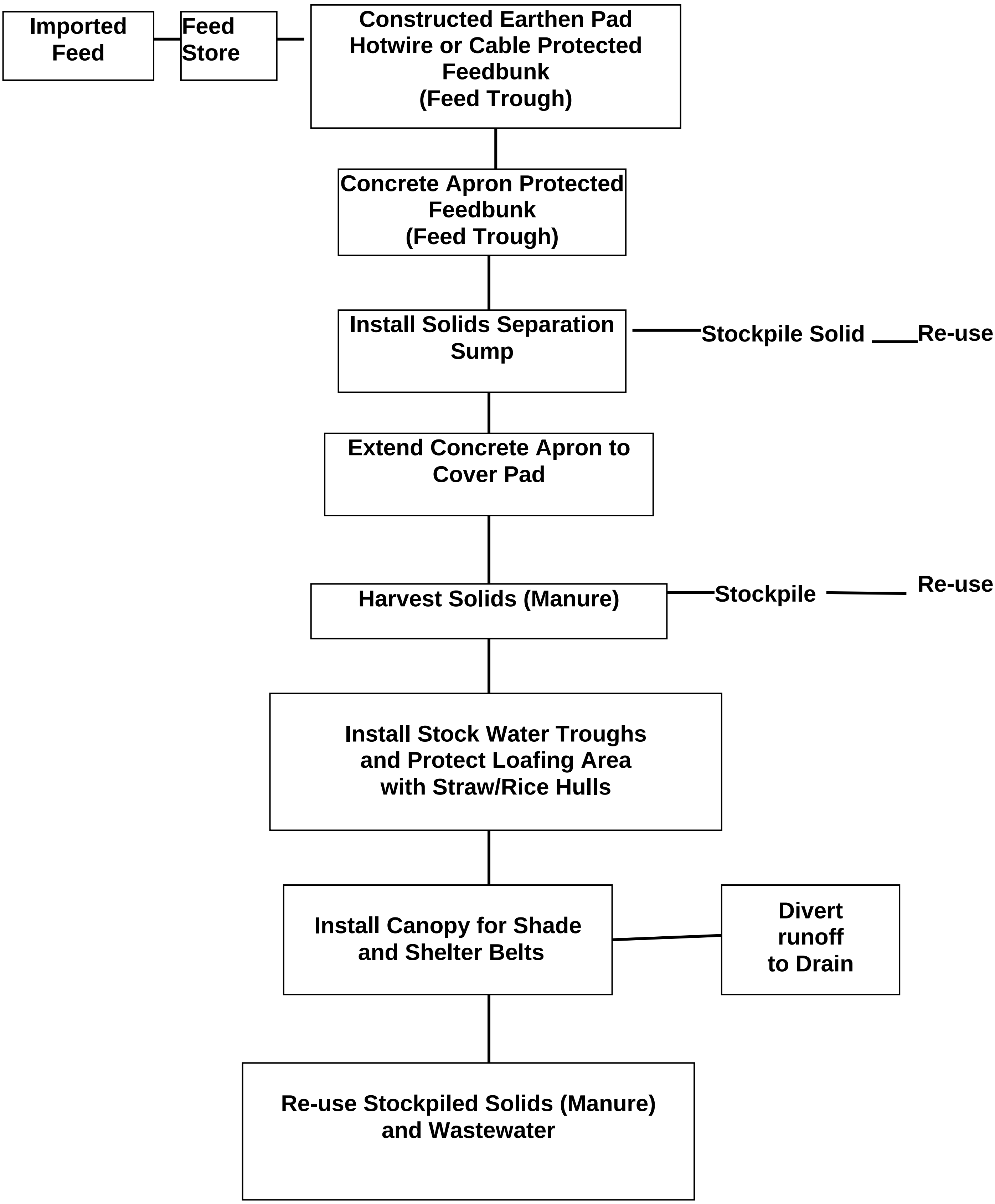
Department of Primary Industry Concerns

- -- animal welfare
- – effluent management
- – use of manure and soiled bedding
- – environmental impact
- – biodiversity.
- – Access to and from feedpad for animals.
- – Vehicle access to feedpad for delivery of feed and removal of manure.
- – Aspect.
- – Shelter and Shade
- – Water Supply.
- – Feed Storage and Baiting for Vermin.
- – Cleaning feedbunk or feed-out area.
- – Control of runoff to and from pad.
- – Control of effluent – storage and re-use.
- – Harvesting and Storage of Manure.
- – Visibility and aesthetics.
- – Stage Construction.

Feedpad Management System



Feedpad Stage Development



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