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).

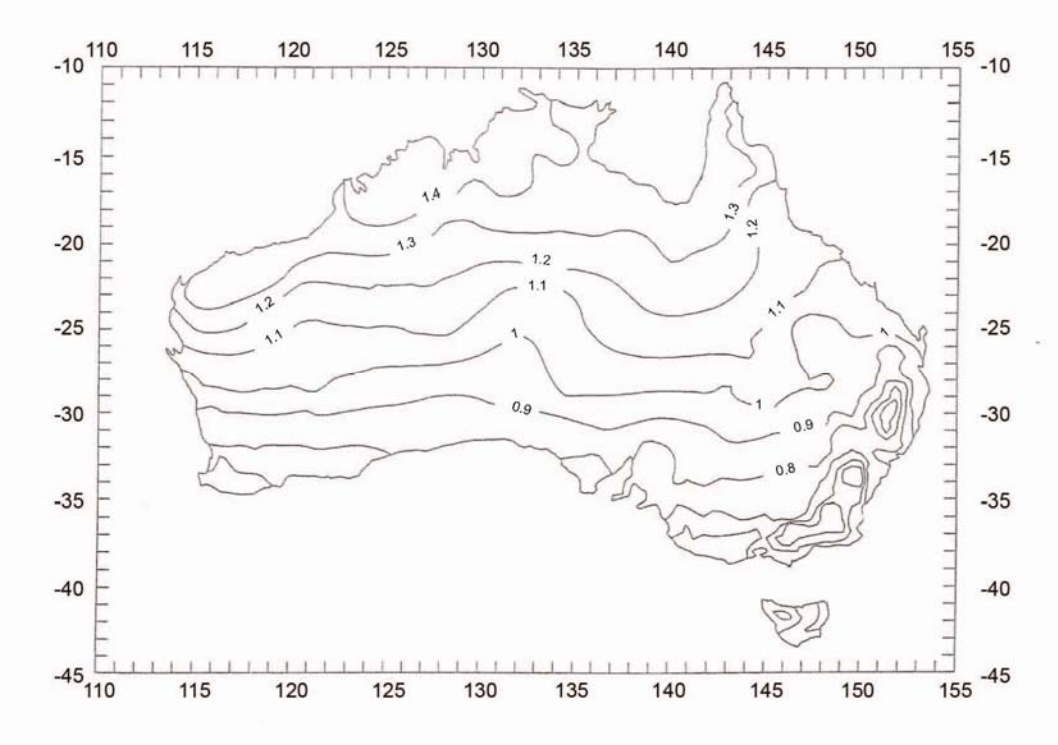

Figure 1. Map of iso-*K* lines in Australia used to adjust $VSLR_{max}$ (Kruger et al. 1995).

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.

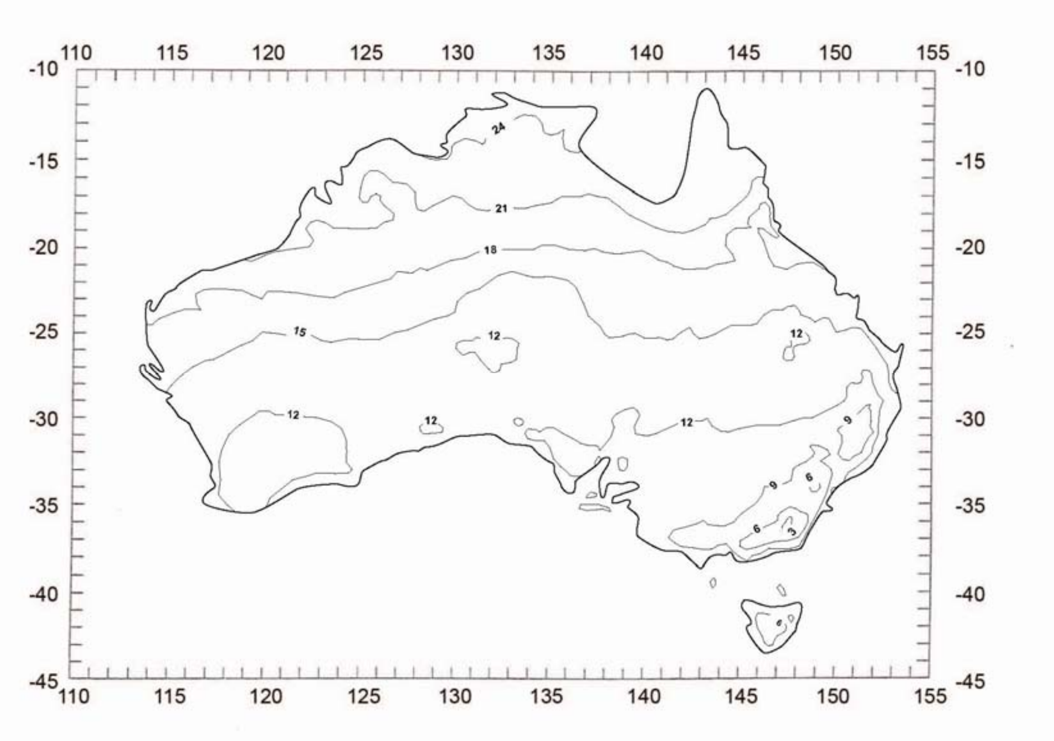

Figure 2. Mean winter temperatures in Australia (source—BOM).

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).

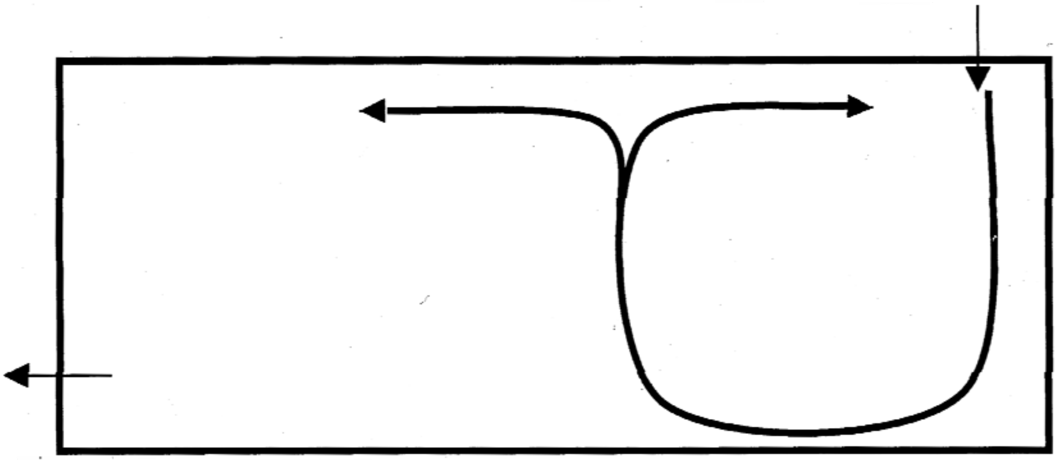

Figure 3. Circulatory currents in ponds with a length-to-width ratio of 3 or more (Shilton and Harrison 2003).

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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