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Dairy Shed Waste Stabilisation Ponds: A review of treatment processes, upgrading options and research needs

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EXECUTIVE SUMMARY

Dairy shed wastewaters in New Zealand have commonly been treated by lagoon systems consisting of an anaerobic pond followed by a facultative (or "aerobic") pond. These pond systems have served New Zealand well, but are now increasingly regarded as insufficient to safeguard the quality of receiving streams and rivers, since high dilutions are required to safely assimilate discharges. The large aggregate investment in existing stabilisation ponds on dairy farms and the continuing need for alternatives to land application, has prompted investigation of options available to enhance pond facilities, so as to better protect receiving waters and meet increased environmental standards.

Contaminants in dairy pond effluent can be ranked according to required dilution factors. The "priority pollutants" are identified as: nutrients promoting nuisance growths, faecal indicator bacteria indicating risk to human and livestock health, ammoniacal nitrogen which can be toxic to stream life and contribute significantly to oxygen depletion, and suspended solids for their effects on stream life and aesthetic quality.

In this report, treatment processes occurring within stabilisation ponds are reviewed and factors limiting the functioning of dairy pond systems are identified. Anaerobic dairy ponds generally work well at their primary function of removing suspended solids and BOD, although some simple measures to reduce sludge carryover to the second pond are recommended. In contrast, the performance of the second, facultative pond, is generally poor, with low oxygen supply apparently limiting treatment performance. Algal populations tend to be relatively sparse and unstable in dairy shed ponds by comparison with otherwise similar domestic sewage pond systems where the majority of oxygen for waste stabilisation is provided by algae. Restricted light penetration in the highly-coloured waters characteristic of dairy wastewaters (euphotic zone restricted to the upper 10–15 cm), and high ammonia concentrations, appear to be the main factors restricting algal growth and also sunlight inactivation of pathogens in these systems.

Various methods for improving dairy pond performance and for providing further treatment are discussed. These approaches fall into three categories: pond modifications (e.g. reducing the depth of the second pond to promote algal growth), pond "add-ins" to existing pond facilities (e.g. mechanical aeration), and "add-ons" to existing pond facilities (e.g. maturation ponds, constructed wetlands or overland flow systems).

Topics for research on understanding and enhancing dairy pond performance are identified and discussed, with emphasis on "low-tech", "passive" technologies that are more likely to be applicable to New Zealand dairy farms than "high-tech", energy- or management-intensive approaches. Key future research directions are identified, including evaluation of the potential of: shallow pond systems (which are expected to develop high algal biomasses and a better-oxygenated wastewater), additional facultative and/or maturation ponds (to provide an extra level of treatment, and buffer against poor effluent quality excursions), simplified overland flow and rock-filter systems (to remove algal solids and to buffer receiving waters from pond effluent), and rotating biological contactors (to enhance biological oxidation of organic compounds and ammonia). Approaches using

1. either mechanical or algal oxygenation to achieve better oxygen supply to pond waters,

2. baffling of ponds (to promote plug flow and more stable effluent quality), and

3. biofilm support structures (to encourage nitrification),

are likely to be worth investigation.

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1. INTRODUCTION

In New Zealand the standard design for stabilisation ponds treating dairy shed wastewater is a two pond system comprising an anaerobic followed by a facultative (originally termed "aerobic") pond. The treated effluent from the facultative pond is commonly discharged into farm drainage systems or streams and rivers. The general design of these ponds was based on guidelines originally produced by the Ministry of Works (MWD, 1972) and reproduced with minor modifications by the Ministry of Agriculture and Fisheries (MAF, 1975; MAF, 1985 developed in consultation with MWD, but withdrawn in 1992), and with loading criteria specified in the Agricultural Wastes Manual (NZAEI, 1984). Certain Regional Councils and Water Boards have produced guidelines based broadly on the above publications (Environment Waikato, 1993).

Studies of effluent quality and/or performance of dairy shed pond systems built to standard design have been reported by Warburton (1983), Dakers (1983), and Hickey et al. (1989). On average the quality of effluents from dairy shed ponds is satisfactory, but there is high variability in effluent quality (e.g., 8-fold between the 5 percentile and 95 percentile in effluent "strength" as measured by biochemical oxygen demand (BOD) values Hickey et al., 1989a), both *between*- and *within*-systems. Differences between average quality of effluents from particular pond systems may reflect poor performance and/or inappropriate waste loadings.

The high variability of effluent quality within any one pond is inherent in a "natural" treatment system subject to the vagaries of the weather. Pond functioning relies on a complex interplay of biochemical processes within the pond. These are strongly influenced by physical processes which vary in response to meteorological forcing. New awareness of the environmental impacts of contaminants in dairy wastewaters other than biochemical oxygen demand and suspended solids (SS), such as ammoniacal nitrogen, phosphorus, pathogenic microorganisms (Hickey et al., 1989), and also the optical effect of effluent constituents (Davies-Colley, 1995), has raised awareness that pond functioning must be improved if receiving streams are not to be overloaded.

Implementation of the Resource Management Act (RMA, 1991) has placed greater responsibility on Regional Councils to address surface water contamination, and has created a demand for increased environmental protection and enforcement of standards. A requirement to fulfil obligations under the Treaty of Waitangi with respect to Maori spiritual values, which generally require wastes to be purified by the earth, has led to surface water disposal being viewed increasingly unfavourably. However, the large investment in stabilisation ponds on dairy farms, as well as the unsuitability of land disposal on certain soil types and the need for careful management even on suitable soils, has prompted a need to find ways of enhancing existing facilities.

There is a large body of research which has been performed on ponds as a method for stabilising organic wastes. Most research has been performed on other wastewaters of faecal origin (especially domestic sewage, but also piggery wastes). Where relevant to dairy shed systems and New Zealand conditions, this research has been reviewed. We also discuss receiving water effects where information is available, particularly from New Zealand research.

In this report we review the present status of waste stabilisation pond technology as applied to dairy shed wastewaters in New Zealand, and discuss some options for enhancement. Specifically, we:

1. Review concepts of waste treatment processes in waste stabilisation ponds
2. Discuss existing dairy waste stabilisation pond systems in New Zealand, identifying problems and potential enhancements, and
3. Identify research needs to improve pond effluents.

PART 1
PROCESSES OF TREATMENT IN WASTE STABILISATION PONDS

2. ANAEROBIC PONDS

Anaerobic ponds are generally 2–5 m deep basins which function to reduce contaminants in organic wastewaters primarily by *settling of solids* (ASAE, 1990). The settled solids accumulate as a layer of sludge on the bottom of the basin, which then undergoes anaerobic digestion. The comparatively small surface area of anaerobic ponds restricts wind mixing and, consequently, oxygenation via surface reaeration. The anaerobic digestion involves a wide variety of micro-organisms hydrolysing the incoming organic matter to produce organic acids, alcohols, sulphides, amino acids and carbon dioxide.

$$\text{Organic matter} + 2H^+ + SO_4^{2-} \rightarrow \text{organic intermediates} + H_2S\uparrow + CO_2\uparrow + H_2O + \text{energy}$$

(Sulphate-reducing bacteria)

$$\text{Organic matter} \rightarrow \text{organic intermediates} + CO_2\uparrow + \text{energy}$$

(Facultative bacteria)

Amino acids are further digested to produce ammoniacal nitrogen (ammonium plus ammonia), sulphide and simple carboxylic acids and alcohols. Methanogenic bacteria convert these acids and alcohols into methane and carbon dioxide. These bacteria require anaerobic conditions and the presence of ammonium as a nitrogen source. Methane can be formed by the reduction of carbon dioxide, or by the decarboxylation of acetic acid (Oswald, 1968).

$$CO_2 + 8H^+ + 8e^- \rightarrow CH_4\uparrow + 2H_2O$$

$$CH_3CO_2H \rightarrow CH_4\uparrow + CO_2\uparrow$$

In principle, anaerobic ponds could be sized so that a balance of sludge digestion and solids sedimentation to form new sludge is achieved. However, sludge accumulation is generally faster than digestion (and entrainment in the effluent) leading to a build-up of sludge level. Therefore anaerobic ponds require periodic desludging. Waste digestion is highly temperature-dependant, with almost no digestion below 10°C and little below 15°C (Hawkes, 1983). As a result, sludge build-up tends to be faster in colder climates.

Formation of methane and carbon dioxide gases (CH_4 and CO_2) during digestion may cause gas-buoyed masses of sludge to rise to the surface. To prevent floating sludge particles from being entrained in the effluent, a T-section may be added to the outflow pipe, and a baffle-board placed under the T-section to divert rising sludge from entering the tee from below. In relatively warm climates, where appreciable sludge digestion occurs, gas-buoyed sludge entrainment in the effluent is greater than in cool climates.

Appropriately-loaded anaerobic ponds usually achieve removal rates of approximately 60-75% of BOD_5 (Mara et al., 1992). Specific guidelines for dairy shed anaerobic ponds in New Zealand (NZAEI, 1984; MAF, 1985) assume 70% reduction, however it appears that in practice, they generally perform better than this, commonly removing around 80% of BOD loads (Warburton and Parkin, 1982; Dakers and Painter, 1983; MAF Policy, 1994).

A considerable amount of research has been devoted to providing appropriate designs for anaerobic ponds, and engineering guidelines for pond construction are well established (ASAE, 1991). Other research has focussed on developing loading criteria, which affects pond size and sustainability. Effective pond operation (60–75% BOD removal) can be achieved with retention times as short as one day in warm climates (Mara et al., 1992). Removal falls to around 40% where average temperatures are less than 10°C requiring increased retention times. Solids accumulation however can be rapid, particularly in colder climates where digestion is slow, leading to a reduction in effective pond volume and retention time, so ponds are generally designed to take account of climatic factors.

Sludge accumulation has been described both in terms of constant annual rates (Ghrabi and Ferchichi, 1994) and decreasing rates (Smith, 1983), with accumulation being most rapid in the first few years of pond operation, then gradually slowing to approach a steady state at which rate of accumulation balances rate of digestion. Various criteria are used for deciding when ponds should be desludged, generally based on the proportion of total volume occupied by sludge. Mara (1987) suggests that anaerobic ponds be emptied when half full, and, based on a Markov modelling approach, Agunwamba (1993) suggests between 60 and 80% full, depending on cost criteria and final effluent quality.

These guidelines assume that sludge accumulation can be easily monitored, however, sludge depth can not be directly visually assessed because of the high light attenuation of the water (if not the presence of a floating "crust" of solids—as occurs fairly frequently on dairy anaerobic ponds). However, sludge level is easily measured with a long stick wrapped in a white towel (Pearson et al., 1987) or various other "sludge probes" (Barth and Kroes, 1985). A boat is usually necessary to access the pond centre.

Loading criteria for anaerobic ponds must take into account the risk of odour production (Ritter, 1989) by reduced compounds such as H_2S. Apparently engineers have been reluctant to use anaerobic ponds for sewage treatment in New Zealand and elsewhere because of their perception that there exists a risk of odour nuisances (Mara and Mills, 1994). Odour production is usually avoided by preventing overloading and by keeping the pond pH > 7.5, so that most of the reduced (–II) sulphur is present as the (odourless) bisulphide ion rather than as H_2S (Mara et al., 1992; Agunwamba, 1993). Excessive organic acid production reduces the pH, which enhances the release of hydrogen sulphide gas, as well as being unfavourable to methanogenic bacteria. Although appropriately-loaded ponds do not generally produce an odour nuisance, odours can be chemically controlled, if and when required, by raising the pH above 9.5, where hydrogen sulphide gas release is negligible because of the shift in the hydrogen sulphide-bisulphide-free sulphide equilibrium away from H_2S (Stumm and Morgan, 1981). Addition of nitrate salts has been suggested for odour control by Agunwamba (1993), the nitrate presumably acting as a favoured oxygen donor over sulphate. Inhibiting anaerobic conditions at the pond surface by maintaining aerobic conditions (Schulz and Barnes, 1990) also prevents odour production, presumably by providing an oxygenated "buffer" region. Physical confinement of odours using pond covers is also used in some instances (e.g. dairy ponds at DRC, Ruakura), and may enhance anaerobic digestion by excluding oxygen and retaining heat.

3. FACULTATIVE PONDS

3.1 BOD and suspended solids removal processes

Facultative ponds function mainly to convert wastewater BOD and suspended solids into algal BOD and suspended solids. In addition, some nutrients are uptaken, some ammoniacal-N may be converted to nitrate, and considerable inactivation of faecal indicator bacteria, and, one presumes, pathogenic microbes, occurs.

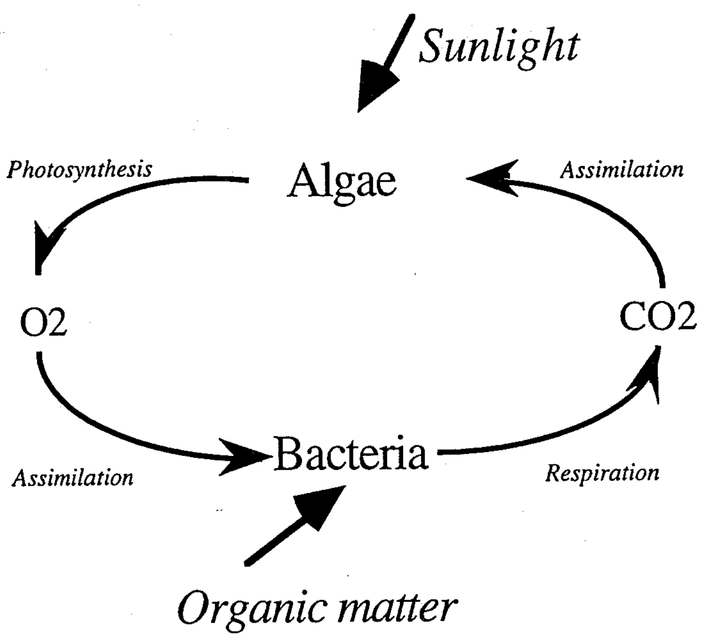
Facultative ponds are designed with large surface areas and shallow depths (1–1.5 m) to maximise sunlight and wind exposure. These ponds are usually located downstream of anaerobic ponds, or are used directly, without an anaerobic step, for comparatively low strength wastewaters such as domestic sewage.

Facultative ponds are characterised by thermal stratification during daylight hours, as the surface layer is warmed by absorption of solar radiation while the lower layer remains cool. The thermal stratification breaks down at night when the surface water has cooled sufficiently by convection and back-radiation, allowing mixing of the two layers (Fig. 1). Sometimes surface cooling occurs at night without mixing, producing an unstable stratification. Stable thermal stratification re-establishes the next morning as the sun again heats the surface water. On windy days, movement of the surface water may inhibit stratification, and in winter, when solar heating of the surface waters is reduced, thermal stratification is comparatively weak. Under either windy or dull conditions, there is less contrast between conditions at the bottom of the pond, and in the surface layer (Gu and Stefan, 1995), although the lower layer will still usually remain anoxic.

High nutrient levels in the incoming wastewater, and CO_2 produced by heterotrophic bacteria from decomposition of organic matter, promote algal growth. However, the highly light-attenuating character of the water reduces light penetration, so restricting algal growth mainly to the surface water layer. Typically algae in properly-functioning facultative ponds achieve populations of algae that are *self-shading* and thus self-limiting. Any increase in algal biomass decreases light penetration, so decreasing algal growth and re-establishing the status quo (Curtis et al., 1994; Davies-Colley, 1995).

Anaerobic digestion may occur in the lower layer of water within facultative ponds, and in the sludge, just as in anaerobic ponds. Bacterial mineralisation of organic matter releases inorganic carbon and other nutrients which become available for algal growth. Oxygen can enter the upper water layer through the surface via atmospheric reaeration, but most oxygen is supplied to the pond water by phytoplankton photosynthesis. During daylight hours, oxygen concentrations in the surface layer can be increased by algal metabolism to levels greatly exceeding saturation. The algae consume CO_2 faster than it can be replenished by bacterial respiration, resulting in a shift of the carbonate/bicarbonate/carbonic acid equilibrium (CO_3^{2-} / HCO_3^-/ H_2CO_3) towards carbonate, increasing the pH of the water. The high pH and dissolved oxygen levels created in the surface waters of ponds with abundant algae, are believed to enhance sunlight inactivation of bacterial indicator organisms such as faecal coliforms and presumably associated pathogens (Curtis et al., 1992), and also the loss of nitrogen by volatilisation of ammonia (Reed, 1985).

The phototrophic and heterotrophic microbial populations in facultative ponds can be regarded as symbiotic (Uhlmann, 1980; Ellis, 1983; Hawkes, 1983), as illustrated overleaf. Bacteria produce inorganic carbon from mineralisation of organic matter, which is used by the phytoplankton for photosynthetic production, in which oxygen is released as a byproduct. This oxygen is used by the heterotrophic bacteria for further mineralisation of organic matter.



Ip et al. (1982) identified light, inorganic carbon, temperature and nitrogen as key variables promoting algal growth. As the main flux of CO_2 for algal photosynthesis in stabilisation ponds occurs through bacterial respiration releasing CO_2 from organic compounds introduced in the wastewater, reduced organic loading or bacterial metabolism may result in carbon limitation of algal productivity (Azov et al., 1982). While the majority of algae are dependent on CO_2 as their carbon source, there have been some reports of facultative heterotrophic utilisation of carbon in the form of glucose (Abeliovich and Weisman, 1978) and water-soluble low molecular weight compounds (Siuda et al., 1991). Such heterotrophy on the part of some algae might be expected to represent a competitive advantage at times of light-limitation in ponds.

Daytime elevation of the pH (resulting from algal CO_2 consumption during photosynthesis), may in addition to feedback effects on algal productivity (via CO_2 limitation and increased ammonia toxicity) also effect the species composition and biomass of algae (Goldman et al., 1982 a; 1982 b) and bacteria (Lallai et al., 1988; Mayo and Noike, 1994) in waste stabilisation ponds. Ip et al. (1982) found that higher temperatures resulted in *lower* algal biomass, which could be explained by reduced CO_2 solubility.

3.2 Nitrogen and phosphorus removal processes

Facultative ponds usually remove a proportion of total nutrients in organic wastewaters. The primary phosphorus removal mechanism is by sedimentation within the pond. Soluble P can be mobilised from the sludge in response to changing redox conditions in the pond bottom waters (Houng and Gloyna, 1984). The overlying aerobic zone in facultative ponds may be significant in phosphorus removal in ponds because of the control oxygen has on the iron cycle. Phosphorus complexes with ferric iron and becomes immobilised in the sludge. Caraco et al. (1989) suggested that sulphate concentration in the overlying water was a critical controlling mechanism of P sorption/release, regardless of oxic/anoxic conditions, but the mechanism was not elucidated.

Wrigley and Toerien (1990) found that removal of soluble N and P in facultative ponds was primarily by biological uptake and either sedimentation, or export as algae in a series of small (4 x 10 m^3) experimental facultative ponds. In monitoring and modelling studies of a full-scale pond system, Ferrara and Avci (1982) also found that sedimentation was the primary means of N removal. In principle, nitrogen removal could also occur via ammonia volatilisation or denitrification, however Ferrara and Avci (1982) found these mechanisms to be inconsequential in the pond system they studied. Pano & Middlebrooks (1982) and Reed (1985) found a significant relationship between ammonia removal and pH, temperature and hydraulic loading rate of the ponds, although neither attempted to identify the causal mechanism beyond suggesting that incorporation within sediments or ammonia volatilisation were the most likely. Nutrient removal can be achieved by harvesting of algae (e.g., for use as livestock feed), as in high-rate algal ponds (Fallowfield and Garrett, 1985), however the high energy and labour requirements would preclude such an approach on dairy farms.

The toxic form of ammoniacal nitrogen is the un-ionised form; ammonia (NH_3) rather than ammonium (NH_4^+). The fraction of ammoniacal-N (=$NH_3 + NH_4^+$) in the un-ionised form is controlled predominantly by pH in the following dissociation equilibrium:

$$NH_4^+ \rightleftharpoons H^+ + NH_{3(aq)}$$

Toxicity of free ammonia to algae has been found in a wide range of algal species (Warren, 1962; Natarajan, 1970; Abeliovich and Azov, 1976; Azov and Goldman, 1982; Wrigley and Toerien, 1990). Free ammonia concentrations of 2.0 mM (28 g m^{-3} NH_3-N) or more have been reported to inhibit photosynthesis. Thus photosynthetic elevation of pH within waste stabilisation ponds is likely to result in auto-inhibition of algae by increased ammonia levels.

3.3 Faecal bacteria removal processes

Pathogen removal is a key role of facultative ponds, so as to ensure that effluents contaminated by faecal wastes do not represent a health risk to humans or livestock. Direct sampling of pathogens is difficult because the range of potential organisms is wide, their occurrence is low and sporadic, and testing for each is expensive. Therefore, ubiquitous, although non-pathogenic, bacterial species are used as an *indicato*r of faecal contamination. Faecal coliform (FC) bacteria are the most commonly used indicator (although presumptive and total coliforms, *Escherichia coli* – a member of the faecal coliform group – and enterococci are also used). Reduction of these faecal indicators in ponds (and other treatment systems) is considered indicative of the reduction of disease risk posed by faecal contamination. Several factors influence the inactivation of faecal indicators in ponds. The main factor is probably sunlight (Moeller and Calkins, 1980; Mayo, 1995), possibly via the formation of photochemical species such as singlet oxygen and superoxide that are highly toxic to bacteria (Curtis et al., 1992). The shorter visible and ultraviolet wavelengths of sunlight are likely to be most important. Penetration of sunlight into pond water may be the main factor limiting solar inactivation because of the highly light-attenuating character of pond effluents.

Faecal coliforms are not ideal indicators of faecal contamination, because, for example, their inactivation rate may be different from that of pathogens, especially viruses, and under some circumstances they are known to multiply in water (Hanes et al., 1964). Higher inactivation rates of faecal indicator bacteria than pathogens may cause an underestimation of potential disease risk. Alternatively, a lower inactivation rate would cause an overestimation—which may be preferable from the environmental perspective. Faecal streptococci, or their subgroup, faecal enterococci, have been suggested as a more appropriate indicator group than faecal coliforms being generally more persistent in the environment (Hanes et al., 1964; Cohen and Shuval, 1972; Fujioka and Narikawa, 1982; Miescier and Cabelli, 1982), although Sinton et al. (1993) have cautioned against use of overseas criteria until there is confirmation under New Zealand conditions. R.J. Davies-Colley and co-workers (pers. com.) have shown that faecal coliforms are *more* persistent than enterococci in sewage ponds at modest pH, and thus may be a preferred indicator of the level of faecal contamination from these effluents.

4. MATURATION PONDS

Maturation ponds are typically smaller and shallower (about 1 m deep) than either anaerobic or facultative ponds, and often two or more are used in series. They are designed primarily to improve the bacteriological quality of the final effluent (Qin et al., 1991), but also function to remove further BOD and oxidize ammoniacal-N to nitrate, and to buffer against excursions towards poor effluent quality. They have a more diverse algal population (Mara et al., 1992) than facultative ponds, and, because of their decreased nutrient supply and algal grazing by protozoa and micro-invertebrates, algal biomass tends to be somewhat lower (Mara et al., 1992). As outlined earlier, faecal indicator bacteria and pathogen inactivation/removal is probably achieved mainly by sunlight exposure in the presence of high dissolved oxygen and high pH.

Under certain circumstances, possibly associated with lowering of ammonia concentrations, aquatic invertebrates can proliferate in maturation ponds, feeding on algae in the incoming discharge from the facultative pond. Because of the increase in clarity as a result of the high grazing pressure on the algae, these are sometimes called "clear-water" ponds (Uhlmann, 1980).

PART 2
DAIRY SHED WASTE STABILISATION PONDS

5. EXISTING SYSTEMS

5.1 General features

There are some important differences between ponds treating domestic sewage (upon which most research has been done) and those treating dairy shed wastewater. Domestic sewage is appreciably weaker in terms of BOD concentration than dairy shed wastewater and also has lower ammoniacal-N and SS levels.

Most of the dissolved organic matter in dairy shed wastes is refractory aquatic humus produced in the cows' gut, which imparts a strong dark yellow colour to the wastewater. In addition, degraded plant pigments such as phaeophytin (the degradation product of chlorophyll) are present in the stabilisation ponds (MAF Policy, 1994 and authors' unpublished data), which are also highly light-absorbing. The resulting strong light attenuation in the dairy shed effluent, restricts the euphotic zone to the upper 15–20 cm of the pond depth (Authors' unpubl. data, Fig. 2B). Domestic sewage ponds are similarly strongly light attenuating, although algae are a greater contributor to light attenuation than humic matter.

Wastewater flows to dairy ponds generally occur in two discrete pulses a day following the morning and evening milkings. These pond systems also receive little waste during the non-milking season (usually winter). The hydrology of dairy ponds is not well understood (in comparison with the composition of discharges), and considerable variation occurs between different systems. Many ponds are known to discharge little or nothing for extended periods during summer, presumably because the combination of high evaporation and seepage loss through (generally unlined) pond bottoms exceeds inflow plus rainfall at that time. The seepage losses from dairy ponds are the subject of current research by Lincoln Environmental (Ray et al., 1995). Impacts of dairy pond discharges upon receiving waters depend not only on contaminant concentrations in the discharge, but also on the dilution factor (and the concentration of contaminants already in the water from other discharges and from diffuse sources), and diurnal and seasonal variations in flow rate. For example, in summer periods streams are generally most vulnerable due to low flow rates (and hence low available dilution levels) and reduced stream channel habitat. These conditions combine with high water temperatures (which reduce oxygen saturation), and increased biological uptake of oxygen and sensitivity of stream organisms to ammonia toxicity. However, low discharges in summer from many dairy ponds may compensate for lower stream assimilative capacity in this season. Conversely, highly pulsed daily discharge patterns may result in receiving waters experiencing very high loadings of pollutants over short periods.

During the winter, waste digestion in the anaerobic pond, and algal growth in the facultative (and any maturation) pond is very limited. However because there is generally little waste entering the pond during this season, the resulting low performance of dairy pond systems may not be so critical. Exceptions include farms producing milk through the winter, or where a portion of the herd is held on the milking pad because the paddocks are too wet to risk stock trampling and pugging damage (MAF, 1980). Neither situation has been much researched. However winter dairy shed discharges occur in a season when receiving water flows are usually relatively high in New Zealand, providing greater dilution of wastes.

It is recommended practice to divert stormwater away from dairy shed stabilisation ponds. Stormwater reduces the retention time of the ponds, and during winter, can reduce residence time and therefore the extent of treatment. This reduces the ability of the ponds to handle the incoming wastes at the beginning of the dairy season, and can result in odour production in the spring.

Anaerobic ponds treating dairy shed wastewater generally operate well, with >70% removal of BOD (MAF Policy, 1994), despite being loaded at 1.4–1.5 times higher than equivalent ponds in USA (ASAE, 1991). Infrequent problems do occur during desludging, when odours are sometimes generated due to anaerobic conditions extending to the surface. Generally however these ponds are fairly resilient to changes, including shock loading with spilt milk or other materials (Warburton and Parkin, 1982; Romli et al., 1994). Larger pond sizing may provide management and economic benefits by reducing the frequency of desludging required.

By contrast, facultative dairy shed ponds in New Zealand are not performing very satisfactorily. Design guidelines for dairy shed ponds (NZAEI, 1984) suggest that facultative ponds should achieve 80% removal of BOD_5, however in practice, facultative ponds appear to be achieving about 40–50% removal on average (Warburton, 1983; MAF Policy, 1994; Mason, 1994). A probable reason for this poor BOD removal is the restricted oxygen supply to the pond water owing to limited algal growth. Restricted oxygen supply also limits oxidation of ammoniacal-N to nitrate (nitrification) in dairy shed ponds.

5.2 Algae

A major difference between dairy shed ponds and sewage facultative ponds is that algal growth in the former appears to be sporadic and generally weak, whereas the latter systems are dominated by algal metabolism. The MAF (MAF Policy, 1994) survey found that some dairy ponds in New Zealand lacked algae, whereas others had appreciable biomass as indicated by algal pigments. The overall picture is that dairy ponds of current standard design are unfavourable environments for algal growth. Growth of algae is encouraged by the high nutrient levels, but inhibited by other factors, particularly poor light penetration and high ammonia. Motile algae, that can optimise their level in the water column and therefore their light environment, are probably dominant (See comparison between domestic and dairy ponds, Fig. 2.A–C).

The value of algae in facultative oxidation ponds treating dairy shed wastewater has been questioned in MAF Policy (1994). Based on a comparison of spot measurements in thirteen pond systems (some with and some without algae) it was suggested that "there is no evidence...that 'aerobic' ponds with algae achieve better treatment in terms of net removal of BOD, SS or ammonia, than ponds without algae". The data presented however (excluding that for Southland ponds, none of which had significant algal populations, and for Northland ponds for which values were not given for both first and second ponds), shows mean ammonia removal of 87% ± 8.6 for ponds with algae, compared to 49% ± 22.3 for those without (both n=4). Mean effluent ammonia-N concentrations (excluding data for Southland) were 35.5 ppm for the ponds with algae compared to 97 ppm for those without.

A better understanding of algal dynamics and the contributions of algal growth to treatment within dairy shed ponds is critical to the development of improved treatment systems. The functioning of facultative pond systems without algae is poorly understood, but we think is likely to be generally inferior to that of ponds with algae unless artificial aeration is employed. Understanding algal dynamics within dairy shed ponds in relation to physico-chemical factors may enhance our ability to maximise the contribution of algae to treatment while minimising their impacts in effluents.

Restricted light penetration is probably the most important factor restricting algal growth in dairy ponds. Euphotic depths measured by the authors averaged about 15 cm, which suggests that average lighting of the whole water column in these ponds is too low to sustain phytoplankton growth (Davies-Colley et al., 1993, pp 113–115). Curtis et al. (1994) have reported that humic matter is the main contributor to light attenuation in domestic sewage ponds, and, similarly, the even higher humic matter ("yellow substance") content of dairy shed wastewater is likely to be responsible for the even more restricted light penetration in these ponds. Removal of biochemically refractory humic matter is difficult, and involves technology probably not feasible or economic at industrial scale on dairy farms. Therefore, the only way to achieve average water column lighting (and algal growth) in dairy ponds similar to that in domestic ponds (in which algae do grow well—Section 3.1) would be to reduce the depth of the dairy ponds, say to around 0.5 m. If facultative dairy ponds were shallower, thereby increasing average light exposure of the water column, algal populations of greater biomass and stability would probably develop. This would improve oxygenation, and consequently enhance BOD and ammoniacal-N reduction. Inactivation of faecal microbes would also be improved—as a result of better oxygenation of the surface water and greater sunlight exposure. A disadvantage of such systems may be the sometimes high concentrations of SS and BOD in the form of algal solids, but these can generally be readily removed using relatively simple supplementary treatment systems such as constructed wetlands.

An alternative strategy for dairy pond design has been suggested by James (1987) and in MAF Policy (1994) based upon the assumption that conversion of organic matter (and BOD) to algal cells does not represent improved effluent quality. In the proposed systems, the second pond is designed and operated as another *anaerobic* pond, attempting to minimise algal development and maximise SS and BOD removal through settling and anaerobic digestion. Shading and wind-sheltering would be facilitated either using artificial covers or surface-floating plants (e.g. duckweed or *Ludwigia*). Deeper ponds could be used for such systems reducing land area requirements. However, in our view, reduction of other pollutants such as ammonia and faecal indicator bacteria, would probably be limited in such systems, and additional treatment would be necessary before safe discharge into receiving waters.

6. TREATMENT REQUIREMENTS FOR DAIRY SHED EFFLUENTS

6.1 General receiving water guidelines for dairy farming catchments

6.1.1 *Oxygen demand*

Dairy shed waste stabilisation ponds were originally designed principally to reduce BOD and SS levels in wastewaters. Mean overall reductions of 90-95 % of BOD and SS are generally achieved in current pond systems, but effluent concentrations are still relatively high (~4-fold higher than those in pond systems treating domestic sewage). In addition, ammoniacal-N levels are very high in dairy pond effluents (~11-fold higher than levels in domestic sewage pond effluents), resulting in nitrogenous BOD (NBOD) levels some 4-fold higher than the measured carbonaceous BOD (CBOD). Maintenance of adequate dissolved oxygen (DO) levels in rivers and streams requires *both* these sources of oxygen demand to be accounted for (Cooper, 1986). Maximum receiving water total BOD (= CBOD + NBOD) levels of 5 g m^{-3} were suggested by Hickey et al. (1989a) so as to maintain DO levels above 5 g m^{-3}. This is the same as guidelines derived from empirical observations of polluted waters in a number of European countries and proposed as an appropriate standard for general use waters under the Water and Soil Conservation Act 1967 (Stevenson, 1980). It is also the value, derived by crude estimation, required to maintain a minimum DO level of 5 g m^{-3}, assuming that atmospheric re-aeration is negligible, the receiving waters are initially saturated with oxygen at 10 g m^{-3}, and that the CBOD and NBOD is exerted within the stream. Re-aeration rates can be very low in lowland streams (e.g. Wilcock et al., 1995), and resident benthic heterotrophs and nitrifiers are often responsible for a large proportion of the oxygen exertion (Cooper, 1986; Hickey, 1988).

The minimum DO value of 5 g m^{-3} used by Hickey et al. (1989a) is the same as the guideline given in USEPA (1976) for maintenance of fish populations (excluding spawning areas for salmonids). More recent criteria (USEPA, 1986) suggest that this level is likely to result in moderate production impairment of salmonids (excluding more sensitive embryo and larval stages) and slight to moderate impairment of non-salmonid fish species. This might be regarded as sufficient protection to avoid a "*significant* adverse effect on aquatic life" (our emphasis) under the RMA 1991 (s. 70) in lowland streams. However, this guideline is unlikely to meet the "stricter" or "more restrictive" requirement in waters managed for aquatic ecosystem purposes (RMA 1991, 3rd schedule) to not allow any contaminant in a discharge to have an "adverse effect". Furthermore, the Third Schedule of the RMA (1991) specifies that DO levels must be maintained above 80% saturation in waters being managed for aquatic ecosystem, fishery, fish spawning or shellfish gathering purposes. This translates to a DO concentration ranging from around ~9 g m^{-3} in winter (10 °C) to 6.6 g m^{-3} in summer (25 °C), or a maximum oxygen depression of only 2.3 to 1.7 g m^{-3} below saturation respectively. In many lowland streams in agricultural catchments such guidelines may be difficult to achieve even in the absence of point-source discharges.

6.1.2 *Suspended solids and light–attenuating materials*

Suspended solids in dairy pond discharges have the potential to reduce the visual clarity of receiving waters and to smother the bed of streams and rivers, changing the nature of the sediments and increasing benthic oxygen demand. There is a lack of information on the optical properties of dairy shed pond effluents, but preliminary data from current studies of 6 Waikato pond systems (authors' unpublished data) suggest that the main factor likely to affect the visual clarity and colour of waters receiving these effluents is the high levels of dissolved humic compounds (yellow substance), rather than SS. The suspended material in dairy ponds seems to have less effect on visual clarity than found for sewage ponds, where small algal cells and algal detritus contribute very strongly to the turbidity and colour. At the time of writing, more data is needed on the optical character of constituents of dairy ponds, to permit an assessment of their likely effects on the colour and clarity of receiving streams.

The SS in discharges from domestic sewage waste stabilisation ponds are mainly of algal origin, but in dairy pond discharges where algal populations are limited, much of the SS appears to be detrital plant material. SS, comprising mainly detrital material from pasture herbage, is likely to have different effects on benthic metabolism than that of less recalcitrant algal solids. There is still debate as to the environmental significance of algal solids discharged from pond treatment systems. Some researchers have argued that algal cells in the discharge from a properly designed and operated stabilisation pond system do not constitute a significant impact on the dissolved oxygen resources of a receiving water, because of their ability to produce oxygen photosynthetically, and be consumed and naturally degraded by stream organisms (Oswald and Ramani, 1976; Gloyna and Tischler, 1981). However, experimental evidence suggests that in many cases pond algae only survive for short periods in natural receiving waters, with respiratory demands and decay resulting in significant oxygen demand (King, 1976; Sutherland, 1981).

Quinn and Hickey (1993) hypothesised that the solids in stabilisation pond effluents may represent either a *subsidy* or a *stress* on receiving water. At high organic solids loads (low effluent dilutions), receiving streams would exhibit signs of stress, with reductions in pollution-sensitive taxa and increases in opportunistic, pollution-tolerant taxa. At lower loads (higher dilutions), pollution-sensitive taxa would not show much effect, while other taxa would increase due to the additional food source supplied. Quinn and Hickey (1993) found partial support for this "subsidy-stress" gradient hypothesis in eight New Zealand streams, receiving discharges from domestic sewage ponds. Dilutions less than 15-fold caused apparent stress (significant changes in the density of > 50% of common taxa and > 50% reduction in the density of sensitive taxa), whereas dilutions of 30 to 50 fold appeared to have caused some subsidy (although this was only statistically-significant at one site). Stressed streams suffered a shift in invertebrate animal assemblages from the pollution-sensitive mayflies and other insects, to pollution-tolerant worms and snails—sometimes occurring at high biomasses. Quinn and Hickey's (1993) results were in reasonable agreement with dilution levels predicted theoretically in an earlier paper (Hickey et al., 1989b) considering effluent quality and receiving water guidelines. Quinn and Hickey (1993) suggested that a final receiving water concentration of ≤ 4 g m^{-3} SS is required to avoid "adverse" ecological impacts in streams. However, as noted by Hickey and Rutherford (1986) the effects of organic point-source discharges are likely to vary depending on the characteristics of the receiving water, with benthic invertebrates in swift stony streams likely to be more sensitive than those in slower-flowing, soft-bottomed streams.

6.1.3 *Nutrients and ammonia*

Nutrient inputs from point-source discharges such as waste stabilisation ponds have the potential to cause nuisance growths of attached algae in receiving streams. Numeric guidelines for dissolved biologically available forms of N and P have been developed to protect contact recreational uses of receiving waters from undesirable growths of benthic algal slimes (MfE, 1992). Although these are the best guidelines available on which to base evaluations of potential pollution risks, New Zealand studies such as those of Welch et al. (1992) and Quinn and Hickey (1993) have shown that, in practice, algal biomass levels in streams receiving domestic sewage pond effluents in New Zealand are often lower than predicted from nutrients alone. Factors inhibiting the development of nuisance growths include high macro-invertebrate grazer densities, riparian shading, and absence of suitable attachment surfaces (e.g. unstable substrates). Because nutrient levels in dairy waste stabilisation ponds are characteristically high, promotion of algal nuisance growths is likely to be common.

However, background levels in agricultural lands are already high, so the additional point source is of less significance than if discharge were to a pristine environment.

Free ammonia is potentially toxic to animals in receiving streams as well as to algae within ponds. Hickey & Vickers (1994) have demonstrated that New Zealand native invertebrates are comparatively sensitive to ammonia by comparison with USEPA (1985) studies. Relative "final acute values" (FAV) found in the New Zealand study, based on the toxicity of 9 native invertebrate species and a native fish species (inanga), were nearly 3.5-fold lower than those proposed by the USEPA (1985) to protect salmonid and other fish and invertebrate species. Surprisingly the most sensitive species are those which would normally be associated with lowland streams, where ammonia-rich discharges from dairy shed treatment ponds would be most common. In the present report, the potential toxicity of ammonia in dairy pond discharges is evaluated in relation to both the accepted USEPA (1985) guideline value (assuming general worst–case receiving water conditions of pH 8 and 20 °C) and a preliminary NZ guideline (assuming the same receiving water conditions) based on the relative FAV value found by Hickey & Vickers (1994) for common New Zealand stream organisms. Ammonia toxicity is most likely to occur at comparatively high pH, where a large proportion of total ammoniacal-N is present as free (toxic) ammonia. The general worst case pH conditions assumed in this report are not likely to be sufficiently stringent for discharges of ammoniacal-N to streams in which high levels of plant photosynthesis and low alkalinity induce more extreme diurnal pH elevations (eg. Manawatu River, Quinn and Gilliland, 1989), requiring application of more stringent guidelines (see USEPA, 1986).

6.1.4 *Pathogens*

The potential disease risk to bathers of contamination of waters by faecal matter of livestock origin is not well known. The only relevant study known to the authors is that carried out in Connecticut by Calderon et al. (1991) who concluded that illness in swimmers was not associated with the level of exposure to animal faecal-contamination of waters. However, McBride (1993) has disputed the statistical interpretation of Calderon et al's study, showing that their data could be interpreted to imply that illness risks from exposure to animal versus human faecal residues are similar. Further research on the human health risk associated with exposure to animal faecal-contamination is certainly warranted, particularly in New Zealand where livestock sources greatly outweigh those from humans. Currently, the specific disease risks to humans from exposure to livestock wastes are necessarily treated as equivalent to discharges from human point-sources.

The present provisional New Zealand microbiological guidelines for recreational freshwaters (DoH, 1992) are based on USEPA standards (USEPA, 1986) and use enterococci and *Escherichia coli* as the indicator organisms of choice. The suggested maximum bathing season (1 December to 1 March) median values of 33 cfu of enterococci or 126 cfu *E. coli* per 100 ml (with various upper limits defined, depending on the degree of usage), are based on assessments of equivalent risk levels to those of the previous USEPA bathing water guidelines based on faecal coliforms (maximum monthly geometric mean of 200 cfu per 100 ml, USEPA, 1976). Considerably lower median faecal coliform levels of 14 cfu per 100 ml (no more than 10% > 43 cfu per 100 ml) are recommended as maximum values for shellfish gathering waters (DoH, 1992). The present report assumes that adherence to the basic microbiological guidelines for full-contact recreation will be the general standard applied to most New Zealand receiving waters, unless other more stringent requirements (e.g. protection of shell-fish resources) are relevant. However, it could be argued that many lowland streams subject to dairy effluent inputs will not be commonly used for bathing or other full-contact recreation and these guidelines may be overly stringent. More relevant guidelines for these situations may be those derived for livestock drinking water, irrigation of field crops, or abstraction for water supply purposes.

The RMA (1991) states (s. 70 and 107) that "no discharge should render a receiving water unsuitable for consumption by farm animals". A wide variety of pathogens and parasites, that may affect livestock growth, morbidity and mortality, are potentially transmissible in faecally–contaminated waters (NTAC, 1968; CCREM, 1987; Calderon et al., 1991). Significant bacterial pathogens (e.g. those causing leptospirosis, salmonellosis, brucellosis and campylobacteriosis) and helminthic parasites, as well as lesser-known sub-clinical bacterial and viral infections that affect stock growth rates and milk production, may potentially be transmitted between farm properties via effluent discharges. However, this is not recognised as a significant problem in the New Zealand dairy industry (pers. comm., Angus Black, Veterinarian, Ruakura Animal Health Laboratory, Hamilton), where dairy cows grazing on pasture are obviously exposed frequently to high levels of faecal contamination, particularly after heavy rainfall and when confined at high densities (e.g. during break-feeding). There appears to have been little research addressing this issue, although Rodenburg (1985) has reported that housed young calves can contract scours (diarrhoea) when exposed to even very low (<5 cfu per 100 mls) total coliform-contaminated drinking water, while older cattle can tolerate higher levels (20-50 cfu per 100 mls) with no obvious ill-effects. There appears to be no specific USEPA guideline for the bacteriological quality of livestock drinking water. Australian and Canadian guidelines for stock drinking waters (CCREM, 1987; ANZECC, 1992) of 1000 cfu per 100 ml are based on a value suggested by Hart, (1974). As the derivation of Hart's guideline is not given and it does not appear to be based on epidemiological studies, its validity may be questioned. The World Health Organisation guideline (WHO, 1989) for irrigation of human food crops eaten uncooked, and for amenity areas such as public parks, is also 1000 cfu per 100 ml for faecal coliforms. No standard is recommended for irrigation of pasture, fodder or cereal crops, where people working the land are exposed.

Protection of the microbiological quality of raw water abstracted for human drinking water supply is another difficult area to define, because modern treatment methods can create drinkable water from almost any water source (CCREM, 1987). The RMA (RMA, 1991) defines "normal" treatment for water supply purposes (third schedule) as the equivalent of coagulation, filtration, and disinfection, but does not give technical specifications for these processes. Previously in New Zealand, the guideline for microbiological quality of raw water for potable supply has been 2000 cfu per 100 ml for faecal coliforms, the value suggested by NTAC (1968) and specified in the Second Schedule of the Water and Soil Conservation Act (1967).

6.2 Priority pollutants in existing pond effluents

Hickey et al. (1989a) have calculated dilution factors required in receiving streams to assimilate dairy shed pond effluents. The dilution factors were calculated by dividing median and 95 percentile concentrations of various contaminants by guidelines for maintaining standards in the Resource Management Act (1991) for streams. Since the volumetric discharge from dairy pond systems may be highly variable, depending on daily milking patterns and weather patterns (and may be non-existent when appreciable seepage or high evaporation occurs), and in-stream assimilation and transformation processes are likely to attenuate contaminants at different rates, the dilution factors calculated in this way must be seen as purely theoretical constructs. However, this approach permits a ranking of contaminants in terms of the required dilution once "reasonable mixing" has occurred in a receiving water. "Priority pollutants" are identified as those requiring high dilution factors.

Recently Davies-Colley (1995) has updated the calculation of required dilution factors given for sewage ponds by Hickey et al. (1989b) using recent scientific understanding and new guidelines. Using a similar approach, Table 1 updates the dilution factors given by Hickey et al. (1989a) for dairy shed ponds. The contaminants for which there is sufficient data for dairy ponds from Hickey et al.'s study are listed in Table 1 in general order of increasing dilution ratio for "safe" discharge. BOD, the contaminant most commonly measured by regulatory agencies in dairy shed pond discharges, is seen to require only around 20-fold dilution to meet basic receiving water guidelines at median concentrations (i.e. protecting receiving waters half of the time) and 50-fold dilution at 95 percentile concentrations (i.e. ensuring protection of receiving waters 95% of the time). However, as discussed previously, nitrogenous BOD imposes a large additional oxygen demand on receiving waters where benthic microbial nitrification occurs. When the full potential oxygen demand (carbonaceous and nitrogenous BOD) is taken into account, required dilution factors rise to 83-fold and over 200-fold for median and 95 percentile concentrations respectively.

Assuming that the criterion (4 g m^{-3}) derived for organic suspended solids in New Zealand streams receiving domestic sewage pond discharges containing mainly algal solids, also applies to the solids in dairy pond effluent (mainly grass fibre, bacteria and some algae), we calculate a required dilution factor of 50-fold to assimilate median SS concentrations to avoid significant impacts on benthic invertebrate communities.

Considerably higher dilutions are required to avoid ammonia toxicity to stream organisms and meet suggested microbiological standards. When the comparatively high sensitivity of New Zealand native invertebrate species (Hickey and Vickers, 1994) is taken into account, required dilution ratios for the 95 percentile concentration increase to nearly 870-fold. Faecal coliforms require dilution ratios of around 540-fold at 95 percentile concentrations to meet basic irrigation and livestock watering guidelines, and 2700-fold to meet guidelines for contact recreation.

The median effluent nutrient concentrations are probably more meaningful for evaluating the effects of effluent nutrient additions than the 95 percentiles—recognising that the response of stream algae to nutrient enrichment is not instantaneous, but depends on average levels available over a comparatively long term. Because the median dilution factors are higher for DIN than for phosphorus, nitrogen may be the more important nutrient controlling nuisance algal growths in streams. Again, it must be recognised that background nutrient levels in streams draining dairy land will generally be relatively high, sometimes exceeding the suggested limiting concentrations, owing to inputs from *diffuse* sources. Thus, point sources such as dairy shed discharges may be less significant overall than would otherwise be the case. In many cases mitigation strategies such as restoration of riparian zones and wetlands, and provision of riparian shading, may also be required to reduce the impacts of nutrients from both diffuse and point-sources in agricultural catchments.

The optical impact (clarity, colour) of dairy shed ponds on receiving streams may be expected to be appreciable, as it is for domestic ponds (Davies-Colley, 1995). However, presently there is not sufficient data on optical characteristics of dairy pond effluent for a meaningful assessment. This is a research task of some urgency, as dairy pond effluent is known from casual observation and recent research (authors' unpublished results) to be highly coloured by humic-type material, as well as highly turbid.

From the dilution factor analysis as summarised in Table 1, the "priority pollutants" are identified as *faecal indicators*, *ammoniacal-N* (as a toxicant, as well as a nutrient and oxygen-demanding substance), *suspended solids* (for their effects on stream life and optics of waters), and where appropriate, *nutrients*. Therefore, upgrading of dairy shed ponds should focus on these contaminants.

7. OPTIONS TO IMPROVE DAIRY POND PERFORMANCE

As a treatment system, dairy shed ponds of existing design can on average achieve substantial removal of BOD, SS, TP and faecal indicator bacteria in dairy shed effluent. However, as noted in Section 5.3, because of the substantial *variability* in pond effluent quality, rather than poor average quality, considerable further improvements may be required before safe discharge can be made to streams (Hickey et al., 1989a) (Table 1). Major improvements can probably be achieved to *existing* pond facilities with comparatively simple modifications or "add-ins". However, there is a limit to the potential removal which can be achieved within ponds of existing design, notably for SS and nutrients, and extra facilities, or "add-ons", may be necessary to achieve further removal of contaminants of concern and, in particular, to reduce the inherent variability of pond effluents.

We recognise that there is a substantial investment in existing pond systems in New Zealand. Therefore, options by which, existing dairy pond systems can be *adapted* or added-to, should be thoroughly investigated before discarding ponds in favour of radically different treatment options. The treatment options discussed here can be characterised (in order of generally increasing cost and complexity) as :

1. "Add-ins" to existing pond facilities (e.g. mechanical aeration, baffling)
2. "Add-ons" to otherwise unchanged existing pond facilities (e.g. maturation ponds, wetlands)
3. Re-designed/configured pond facilities (e.g. increased pond size), and
4. Alternative processes of treatment (ponds discarded)

In the present report we will discuss mainly "add-ins" and "add-ons", and look briefly at the evidence for increasing the sizing of ponds. These approaches are summarised in Table 2. A range of alternative treatment options for dairy shed wastewaters have been discussed and evaluated by Cooke et al. (1992), and Waste Solutions Ltd. are currently investigating the potential of simple activated sludge systems for dairy shed effluent.

Most of the options discussed below are shown schematically in Fig. 3 to illustrate concepts and layout of facilities. The different options are illustrated *individually* (with a few exceptions), although in practice many upgrading options would be combined. For example, shallow pond systems, which are expected to be well-oxygenated owing to improved growth of algae, would probably best be combined with constructed wetlands, overland flow or other means for removal of algal solids before discharge to receiving waters. Biofilm supports and baffles designed to enhance performance of existing facultative ponds would probably best be used in association with mechanical aeration.

7.1 Anaerobic Ponds

Anaerobic ponds are designed primarily for SS removal, but often removal of trapped solids ("desludging") is not planned for at the time of construction. The standard length to width ratio of <2:1 has, in practice, resulted in many dairy anaerobic ponds being almost equidimensional (1:1 ratio). As herd size increases, maintaining this ratio has resulted in an increase in the distance from the edge of the pond to the centre. Standard "suck-and-blow" desludging equipment may not be able to reach all of the sludge on the bottom of wide ponds. As a result, many large ponds receive only partial desludging. If desludging is required more frequently because it is inefficient, this represents an additional cost to farmers. A maximum pond width may need to be specified, or else modifications made to desludging equipment, to permit more effective sludge removal. Based on recommendations in MAF (MAF Policy, 1994), DEC (in prep.) have suggested maintaining length to width ratios at ~ 2:1 or greater (to reduce short-circuiting) with a maximum width of 24 m (to facilitate desludging), with the long axis of the pond orientated perpendicular to the prevailing wind (to reduce mixing).

Despite the ease of measurement, sludge level appears to be infrequently assessed in N.Z. dairy ponds, and desludging interval is often based on arbitrary time intervals (eg. 5 years; Eddy Grogan, ARC, pers. comm.). Rather than attempting to specify "universal" de-sludging intervals for anaerobic dairy ponds, we favour a simple empirical approach based on *monitoring* of the sludge volume. Measurements of sludge depth should be made by regional council inspectors or their agents (or by farmers) with "sludge probes" such as a white towel-wrapped pole (Pearson et al., 1987). A boat or simple (but stable) raft will normally be required to access the pond centre. Desludging could be undertaken following first recognition that the sludge volume was greater than, for instance, two thirds of the pond volume (NZAEI, 1984). A campaign to educate dairy farmers about the need for monitoring, and use of simple techniques to assess sludge depth, would promote adoption of more cost-effective desludging intervals and reduce the incidence of sludge carry-over from ponds with excessive sludge build-up. Such monitoring would also allow farmers to judge the effectiveness of pond desludging operations.

MAF Policy (1994) has noted the benefits of fitting baffle boards under the outlet "T" section in anaerobic ponds, to avoid entrainment of rising, gas-buoyed sludge and possible blockage of outlet pipes (Fig. 4). Discussions with farmers by the authors suggest that use of such a device is the exception rather than the rule. A similar effect may be obtained from use of a second "T" added to the first so that pond water is sucked *laterally* into the outlet piping (Fig. 4). These simple measures should reduce suspended solids carry-over to the facultative (secondary) pond, although it is uncertain how much of a problem this is and what effect it has on overall pond performance.

Research Need:
An appreciable benefit is expected from these measures, but the research need is minimal. Implementation would be enhanced by providing clear specification of optimal design and operational requirements and explaining the rationale behind them.

7.2 Facultative ponds— modifications and "add-ins"

7.2.1 *Increased pond size*

Hickey et al. (1989a) were not able to detect any trend in performance of 11 dairy pond systems with pond size, even though the ponds studied ranged up to 2.3 times the recommended size for anaerobic ponds and up to twice the size for the aerobic (facultative) ponds. A similar lack of apparent relationship between pond size and performance has also been reported for sewage pond systems (Hickey et al., 1989b; Davies-Colley et al., 1995). There is thus little evidence that increasing the size of present "aerobic" ponds will significantly improve discharge quality, and it is unlikely that increased sizing can overcome key factors limiting their performance (i.e. high light attenuation by dissolved humic compounds inhibiting algal growth and restricting oxygenation). Recent unpublished data (Environment Waikato and MAF Policy 1994) has suggested that the estimates of waste production from dairy sheds used in the MAF (1975; 1985) and NZAEI (1984) pond guidelines may be lower than typical, resulting in pond sizes being too small to effectively treat the waste loads produced. This has lead to suggested increases in pond sizes (Figure 3, option 2) (DEC, in prep.) based on revised cow waste load estimates (increased from 90 to 120 g cow^{-1} d^{-1}; MAF Policy, 1994). The changes in loading suggested would lead to an approximate 33% increase in the size of the anaerobic pond and 44% increase in the "aerobic" pond.

Research Need:
The performance of larger-sized ponds as advocated in industry guidelines (DEC, in prep.) should be evaluated and compared with that of existing size pond systems. Because of the inherent variability in between-pond performance, this may best be done using a large-scale experiment set-up that allows ponds of different size treating the same wastewater to be compared side by side. Studies might need to be carried out at a number of sites to ensure generality.

Extensive datasets from presently unpublished monitoring studies of dairy pond systems, such as those of the Auckland and Waikato Regional Councils, should also be investigated to evaluate relationships between pond size and performance, and identify factors affecting the performance of individual pond systems. A range of interrelated factors are hypothesised to be limiting the performance of existing dairy pond systems. These hypotheses need to be investigated and the relative importance of different factors determined.

Relationships between algal abundance, key physico-chemical variables and treatment performance should be determined for a range of ponds of standard design. Data from these studies could then be used in future modelling studies to guide development of alternative pond designs. Accurate in and out-flow monitoring is required for a representative series of dairy shed ponds of standard design. This should be related to shed wash-down operations, pond seepage and transpiration losses, and rainfall inputs. This data could then be used in association with information on pond pollutant concentrations, to provide improved predictions of the treatment level improvements required to minimise impacts on receiving waters.

Studies of effluent quality, treatment processes, and water balances in dairy shed ponds of existing design are the subject of ongoing research at NIWA, Hamilton.

7.2.2 *Shed cleaning and herd management practices*

Dairy shed waste production is likely to be primarily related to the *time* each cow spends in the yard on average. Waste production may thus be reduced where cows are retained for shorter periods of time in the yards; for instance, where there is a high ratio of milking stalls to cow numbers, or where herds are divided into smaller groups that are milked separately rather than kept in the yard as one large herd. There is presently little information on the potential practical and environmental implications of such options and no clear guidelines on these issues.

Present shed cleaning practices employing high pressure hoses generally result in the production of around 50 L of wastewater per cow each day. Volumes in excess of 90 L per day may however occur where more intensive washing regimes for the milking machinery are operated (NZAEI, 1984). Reduction of water use during the shed cleaning process is likely to improve the treatment efficiency of existing pond systems, by reducing dilution of the wastes and providing increased retention times. In many cases dairy shed water use could be significantly reduced if farmers were aware of the issue. More attention to cleaning practices, yard design and stormwater management are likely to also be important. Alternatively, use of different yard cleaning techniques and equipment (e.g., blade-scrapers fitted to the base of the backing gate, and steel grate-covered drainage channels set in the milking pad to assist washing down) may reduce water usage.

Research Need:
Practical means of reducing waste loads during milking operations should be investigated, and appropriate guidelines developed, based on scientific and practical evaluation.

7.2.3 *Inlet and outlet structures*

Uhlmann (1980) recommends that the inflow pipe should discharge effluent from the anaerobic pond into the facultative pond *below* any thermocline so as to minimise channelling directly to the outflow, a phenomenon known as "surface streaming" (Ellis, 1983). NZAEI (1984) recommend that discharge of the anaerobic pond effluent be 300 mm below the surface of the facultative pond, while Mara et al. (1992) specified a greater depth of 750 mm in recent guidelines for sewage ponds. In the absence of specific information on the vertical flow characteristics of influents, we favour adoption of a comparatively deep discharge level, eg. 500mm, to ensure that channelling of inflows in the epilimnion is minimised.

Pipework to and from existing facultative dairy ponds are often inappropriately positioned, with potential short circuits resulting from the outlet pipe not being positioned diametrically opposite the inlet structure (e.g. at opposite corners of the pond). This problem is relatively easily corrected by modifying existing pipework.

Outflow structures in dairy ponds come in many different designs. It is recognised that they should not be at the water surface, as this can result in carry-over of floating solids. Surface takeoff structures are also prone to blockage. Conversely if the off-take is too deep, then deoxygenated water from the lower layer (hypolimnion) will be discharged, with possible adverse effects on aquatic organisms in receiving waters. Reduced algal carryover helps maintain the algal population in the pond (and therefore its effective treatment potential), as well as reducing algal-derived BOD and SS loads on receiving waters. Many of the species of algae found in ponds are motile and exhibit phototaxis, that is, they move towards (or away from) light so as to optimise their light environment. In the highly light-attenuating environment of dairy shed ponds, algae tend to occur within a few centimetres of the water surface, unless wind mixing distributes them through the water column. Outflow structures should also be sufficiently low to avoid entraining algae. Therefore outflow structures should ideally be designed to restrict algal and sludge carry-over in the effluent by withdrawing pond water from *beneath the euphotic zone* (see glossary), but *above the thermocline*. Outflow pipes that withdraw from below the euphotic zone, but still in the epilimnion (~300 mm below the surface) are specified in both MAF and NZAEI guidelines (NZAEI, 1984; MAF, 1985). "T" sections on outflow pipes are a particularly simple and effective means for controlling off-take depth (Figure 4). Reverse slope overflow pipe designs are more prone to blockages, as well as more difficult to accurately position to achieve the desired off-take depth. Generally outflow structures can be readily improved at little cost.

Even with a T-section at an appropriate depth, some algae will still be entrained in the effluent. In principle, it may be possible to reduce the entrainment of *motile* algae may be able to be reduced during relatively calm conditions, by shading a small area of pond water in the vicinity of the outflow pipework. As far as we are aware, this simple approach has not been tried. The size and extent of the shaded area required to ensure that algae are sufficiently far from the pipe to avoid entrainment in the discharge current, would need to be investigated, as well as the wind conditions under which it operated effectively. A reduction of algal concentration in the outflow will induce a nearly proportional reduction in the SS and BOD in the discharge.

Research Need:
Optimising inlet and outlet structures, especially takeoff depths, has the potential to substantially improve effluent quality and possibly pond functioning. There is probably little requirement for research on many of the relatively simple modifications suggested above, but the potential to reduce entrainment of motile algae by shading of the outlet zone would seem worthy of at least preliminary investigation. An experimental approach to examine algal dynamics in dairy shed ponds from the perspective of optimising treatment and effluent quality, would seem to be particularly valuable, enabling for instance, outlet take-off depths to be optimised to reduce algal entrainment in the discharge.

7.2.4 *Shallow depth ponds*

As we have seen, light penetration into wastewater in dairy shed ponds of current design, is restricted by high light attenuation. Restricted light penetration and low oxygen levels reduce photoinactivation of faecal indicator bacteria, and inhibit the formation of an extensive population of nitrifying bacteria (which require an attachment surface in the aerobic zone) leading to poor conversion of ammoniacal-N to nitrate.

Simply lowering the water level of existing facultative ponds, by reconstruction of the outlet pipework, may serve to greatly improve the light exposure and algal growth, with consequent improvements in oxygen supply to the wastewater, and hence BOD conversion and nitrification of ammoniacal-N (Figure 3, option 3). The pond depth would probably have to be lowered to about 0.5 m (roughly half existing pond depth) to provide adequate lighting. This is based on the rule-of-thumb that if the ratio of mixing depth to euphotic depth exceeds 5, the algae will be light-limited (Davies-Colley et al., 1993, pp 113–115). High rate algal ponds (HRAP) which combine shallow depth with mechanical aeration (Fallowfield and Garrett, 1985) are a proven technology overseas for treatment of piggery wastes and other high strength organic wastes. Recent studies of the performance of parallel facultative and maturation ponds in an extensive experimental pond complex treating sewage (Pearson et al., 1995; Silva et al., 1995), have shown that their performance is primarily a function of surface area, with equivalent or improved removal of BOD, N and faecal coliforms in shallow systems operated at the same areal loadings (despite lower residence times because of the lessened volume).

Research Need:
Reducing the depth of existing facultative ponds (and consequently reducing residence time) is an apparently radical concept, but one which would seem to have high potential for major effluent quality improvement at rather low cost. We recommend this as a research priority. Modelling of algal growth response, as well as pilot scale experiments and full-scale trials, should be conducted in tandem in a co-ordinated programme of studies to prove or otherwise, the validity of this concept. Consideration will have to be given to add-ons that can sustainably remove algal solids, such as rock filters, overland flow systems and wetlands.

7.2.5 *De-sludging*

Sludge may build up in facultative ponds as well as anaerobic ponds (Ghrabi and Ferchichi, 1994). In dairy ponds, the sludge probably comprises mainly undigested grass fibres and sedimented algal solids. During anaerobic pond desludging, some farmers also partially desludge their facultative pond, by using water pumped from the facultative pond to stir up the thick sludge deposits at the base of the anaerobic pond. However, the necessity for desludging facultative ponds and its potential benefits are not well-established. Facultative pond sludge comprises both algal detritus and "carried over" anaerobic pond sludge (which should be minimised by off-take baffles or double "T's" and appropriate anaerobic pond desludging). Excessive build-up of sludge in the facultative pond creates a large reservoir of oxygen demand at the bottom of the pond which probably contributes to their relatively low level of performance.

Research Need:
Low. Some research by way of a low-level survey is suggested, involving measurement of the sludge levels in a range of facultative dairy ponds so as to scope the extent and speed of sludge build-up. Consideration of facultative pond desludging may become important if dairy ponds are to be shallower so as to improve algal growth—because a shallower depth necessarily reduces sludge storage volume and increases the likelihood of resuspension.

7.2.6 *Mechanical aeration*

Mechanical aeration (Figure 3, option 4) is widely used, both in New Zealand and overseas, to upgrade and enhance treatment in facultative ponds and to reduce production of nuisance odours (Rich, 1980; Reed et al., 1988; Koottatep et al., 1993) by eliminating anaerobic zones. It is routinely employed for "strong" organic wastewaters such as those from piggeries (Williams et al., 1989; Schulz and Barnes, 1990), wool scourers and meat works. Mechanical aerators would seem to have good applicability to present dairy shed pond systems, because they should overcome the restricted oxygenation and give a more consistent effluent quality (less dependent on weather).

To put mechanical aeration in perspective, it is useful to compare the potential oxygen supply by aerators with that from algal photosynthesis. Air has an oxygen concentration of approximately 20%. Mechanical mixing of this air with water in a pond can only increase the dissolved oxygen to atmospheric equilibrium, while algal photosynthesis can produce oxygen supersaturation with respect to air. However algae cannot photosynthesise at night, and during the day algae in dairy ponds may reside near the water surface, with low oxygenation of bottom waters.

Mechanical aerators cause considerable mixing of the water column, so that temperature, oxygen concentration and pH are fairly consistent from the surface down to the water–sludge interface (Fig. 2C). Oxygenation of the sludge surface may promote development of an extensive nitrifying bacterial population, with appreciable nitrification of ammonia. Mechanical aerators mix the algae throughout the water column, reducing their average lighting. This reduces photosynthetic oxygen production and the pH elevation caused by algal uptake of carbon dioxide. Typically partial mechanical aeration will reduce, if not actually extinguish, algal growth, so that oxygenation by algal photosynthesis is replaced by atmospheric oxygen supply. Inactivation of faecal indicator bacteria may also be reduced because of lowered pH and average DO in the surface layer.

Research on the potential improvement in dairy effluent quality by mechanical aeration is being carried out by NIWA with funding from the Dairy Research Institute. Results from the first phase of the study (Sukias et al., 1995) have been very encouraging. Treatment was compared over a 6 month period under partially aerated and unaerated (control) conditions in a large pond divided and reconfigured to provide two ponds of equal size, operating under the same climatic conditions and receiving the same wastewater loadings. Significant improvements in total ammonia and residual oxygen demand were noted after an approximate six week initial establishment phase, as well as reduced variability in effluent quality, including faecal indicator bacteria.

Various mechanical aeration strategies are possible. The simplest is *continuous* aeration, which may be "full" or "partial", depending on whether there is sufficient aeration capacity to maintain the sludge completely in suspension or permit this to settle within the basin. An alternative strategy is *intermittent* aeration (Araki et al., 1990; Nakajima and Kaneko, 1991) to promote N-removal by sequential nitrification and denitrification. During the aeration phase, nitrifying bacteria oxidize ammonia to nitrate. During the non-aerated phase, oxygen falls rapidly due to metabolism by heterotrophic bacteria, resulting in anoxic conditions. Nitrate is then used as the next most thermodynamically-favoured oxidant after molecular oxygen, by facultative denitrifiers (providing sufficient organic carbon sources are available), and in the process is reduced to dinitrogen and nitrous oxide gases (denitrification).

A wide range of intermittent aeration periods and frequencies are possible (Araki et al., 1990), although high frequencies of on/off cycles may accelerate wear of aeration equipment. Koottatep et al. (1993) have examined the simple strategy of intermittent aeration during the day, and no aeration during the night for sewage, which gave good BOD and TN removal. NIWA is currently trialing *night-time* mechanical aeration for dairy shed ponds, in a second phase of studies being carried out for the DRI using the split pond facility used earlier by Sukias et al. (1995). Night-time aeration is hypothesised to allow stratification to develop during daytime, with algal photosynthesis potentially providing supersaturation of oxygen and high pH values near the water surface, both of which should enhance faecal indicator inactivation. With such an aeration regime, stratification during daylight hours should permit anaerobic conditions to develop at the sludge surface, so encouraging denitrification.

Use of mechanical aeration for odour control is generally not necessary for dairy shed (facultative) ponds as nuisance odours tend to be infrequent and of low strength (and neighbours remote). However odours were an important issue for a dairy shed pond at the Dairying Research Corporation, which was situated close to residential suburbs in Hamilton (Sukias, 1995). The problem was successfully treated by mechanical aeration, with the additional benefit of considerable nitrification of ammonia in the wastewater.

Research need:
Research by NIWA for the DRI is continuing on the potential of mechanical aeration of dairy wastewaters. Investigations are now focused on comparative evaluation of continuous and night-time only aeration of dairy ponds (intermittent aeration). Supplementary studies are required to identify the key zones (planktonic, benthic, pond margins or some combination) where nitrification is occurring, and provide information on *in-situ* rates and the factors influencing nitrification and denitrification. This information could then be used to optimise the efficiency of mechanical aeration regimes, and to develop guidelines that will facilitate appropriate implementation within the dairy farming industry.

7.2.7 *Biofilm attachment surfaces*

Geotextile surfaces used as a site for biofilm attachment, have been shown to enhance removal of pollutants, especially ammonia, from pond wastewaters in a number of studies. Combining mechanical aeration with the use of geotextile support surfaces for biofilm development, can considerably enhance nitrification in pond systems (Constable et al., 1989; Valentis and Lesavre, 1990) (Figure 3, option 5). If the biofilm is close to the water surface, it is likely to contain algae as well as bacteria. Although attached algae (unlike planktonic algae) may create elevated oxygen concentrations in close proximity to bacteria, including nitrifiers, during daylight hours, oxygen usage by attached growth biofilms generally exceeds production in high strength wastewaters. Mechanical aeration provides supplementary oxygen supply, while mixing enhances the supply of substrates to the biofilm, and encourages sloughing of excess biomass and accumulated particulates.

Constable et al. (1989) recorded nitrification rates in a domestic sewage pond of 30 to 40 mg N m^{-2} hr^{-1} on a variety of different support materials such as fishing net, pieces of car tyre and PVC sheeting. Baskaran et al. (1992) recorded up to 30% removal of ammonia (attributed to nitrification/denitrification) in tanks containing biofilm-coated plates fed with domestic sewage. Shin and Polprasert (1987; 1988) found that attached-growths enhanced COD, total phosphorus and suspended solids removal in sewage ponds, but attributed ammonia removal in their studies mainly to biological uptake and incorporation within the biofilm. Valentis and Lesavre (1990) demonstrated enhanced BOD_5, COD and TSS (total suspended solids) removal in a similar attached-growth system treating an industrial organic wastewater, where back-flushing was used to enhance performance by removing attached solids.

We expect that biofilm supports would enhance nitrification in dairy ponds of existing design if combined with mechanical aeration (Fig 3, 5). Nitrification is otherwise likely to be limited in dairy ponds of existing design because of their restricted oxygen supply. The performance of shallow ponds, which are expected to be better oxygenated owing to increased algal metabolism, may also be enhanced by the use of biofilm supports.

Research need:
Biofilm supports are worth investigating in dairy shed ponds in combination with aeration and other enhancements. Initially the heterotrophic, nitrification and denitrification capacities of biofilms that develop in existing ponds could be determined. This could be achieved using laboratory bioassays on small intact sections of biofilm that have been suspended at various positions and depths in aerated and unaerated ponds. The biofilm capacities could then be used, along with models such as that of Polprasert and Agarwalla (1995), to evaluate the potential of biofilm supports and guide field-scale trials.

7.2.8 *Baffling or multiple ponds*

Hydraulic flow characteristics of ponds can substantially affect their ability to treat wastes. Flow patterns are affected by wind and thermal stratification as well as pond shape, the presence of dead spaces, existence of density currents, and by inlet–outlet configurations (Middlebrooks et al., 1982) which, along with mixing, reduce the effective residence time of wastewater in a pond.

Plug-flow systems are likely to provide better removal of contaminants with high rate coefficients (such as bacteria) than completely-mixed systems (Juanico, 1991). For wastes with low removal rate coefficients such as BOD, fully mixed and plug-flow systems are expected to give similar levels of treatment. Baffling of ponds can be used to induce plug-flow where improved removal of faecal microbes is required (Figure 3, option 6a). Careful consideration must be made of inlet-outlet configuration, as short-circuiting within layers during daytime thermal stratification can still occur in baffled ponds, potentially nullifying improvements which might otherwise be achieved (Pedahzur et al., 1993). Kilani and Ogunrombi (1984) reported gradually increasing effective detention times, and a near doubling of BOD and solids removal in laboratory-scale ponds as they increased the number of baffles from zero to 9. Some of the improved treatment recorded in these experiments might have been due to the increased surfaces for biofilm growth provided by baffles. In laboratory-scale experiments comparing 3 different baffle configurations with unbaffled control ponds, Reynolds et al (1975) found highest organic removal efficiencies in ponds with parallel longitudinal baffles which maximised plug flow.

Not all research into plug flow systems have found improved treatment however. Recent comparisons of facultative pond performance in pilot-scale pond systems (Pearson et al., 1995; Silva et al., 1995), found little difference in BOD, COD, SS, total and ammonia-N, or faecal coliforms from domestic sewage, with length to width ratios ranging from 1:1 to 6:1. In these experiments Salmonellae removal appeared to be slightly better in the elongated pond, but rotovirus removal was marginally poorer.

Theoretically baffling of existing dairy shed facultative ponds could substantially improve bacteriological quality of dairy shed wastewaters. In instances where total pond area is sufficient and pond shape is suitable, it may be preferable to divide existing large dairy ponds to form multiple pond systems rather than baffling (Figure 3, option 6b). Multiple pond systems can also greatly improve treatment, notably for faecal microbes, for much the same reasons that plug flow improves treatment. The newly-created pond that first receives the anaerobic effluent should not be overloaded, which might give rise to algal ammonia toxicity, or excessive oxygen consumption.

Like biofilm supports, baffling (or multiple ponds) would seem to be an enhancement which would best be combined with changes to improve the oxygenation of dairy shed wastewater.

Research need:
Not a priority research need in the short-term, but definitely worth investigating in combination with other enhancements in the future.

7.3 Supplementary treatment—"add-ons"

A review of systems for reducing ammoniacal-N from rural point-sources discharges, including potential "add-on's" to ponds, has been compiled by Cooke et al. (1992). Many of the systems considered have the potential to substantially enhance discharges from dairy shed facultative ponds, however maximum performance from each stage in a treatment system depends upon adequate performance from preceding stages. That is, *stabilisation pond performance must usually be maximised to allow add-on's to operate effectively.*

The report by Cooke et al. (1992) ranked systems for ammonia reduction according to technical feasibility, natural environment impacts, human environment impacts, and cost. The systems considered are listed below in order from highest to lowest ranking. Many of these systems would appear at first sight to be too "high tech" and operator-dependent to be suitable on dairy farms, with the exception of the first four (highest ranked) systems.

Overland Flow
Constructed Wetlands
Land Irrigation
Rotating Biological Contactors
Zeolite Beds (Biological regeneration)
Aerated Oxidation Ditches

Sequencing Batch Reactors
Fixed biofilm Reactors
Modified Activated Sludge
Zeolite Beds (Chemical regeneration)
Assimilative Ponds

The first four treatment options noted above will be discussed as well as two relatively simple options not reviewed by Cooke et al. (1992); maturation ponds, and rock and sand filters. In respect to the remaining options not discussed further in this report, NIWA is undertaking FRST-funded generic research on the applicability of NZ zeolite minerals for ammonia removal from wastewaters, and Waste Solutions Ltd is evaluating the use of simplified activated sludge systems for treatment of dairy shed wastewaters under contract to the Dairy Research Institute.

7.3.1 *Maturation ponds*

Multiple-pond systems can greatly improve effluent treatment. For domestic waste treatment overseas (although not commonly in New Zealand), one or more maturation ponds (Figure 3, option 7) typically follow the facultative pond (Mara, 1987; Mara and Marecos do Monte, 1990; Mara et al., 1992). Marais (1974) found that using four equal-sized ponds, each having a 2.5 day retention time (total 10 days) achieved 3 decade reduction of faecal coliforms (0.1% remaining), whereas a single pond of 10 days retention time removed only 95% (5% remaining). Thus, multi-pond systems are generally capable of producing much better quality effluent than 1 or 2 pond systems, and effluent of a superior average bacteriological quality to most mechanical plant alternatives (Price et al., 1995), and can also give fairly consistent effluent quality (reduced variability).

Addition of maturation ponds to existing two pond dairy systems could be relatively easily achieved in many instances. Such ponds would need to be appreciably shallower than standard facultative ponds if algal growth and oxygenation, as well as sunlight exposure for disinfection, were to be optimised. If sufficient algal growth can be encouraged, extensive nitrification should occur in properly functioning maturation ponds.

Research need:
Maturation ponds should be investigated in combination with modelling and pilot-scale studies on shallow facultative ponds to determine the optimum pond configuration, and their performance treating dairy shed wastewaters.

7.3.2 *Rock and sand filters*

Rock filters (Figure 3, option 8) and back-flushable sand filters can remove a considerable proportion of SS and BOD_5 associated with algae in pond discharges (Middlebrooks et al., 1982; Ellis, 1983;

Middlebrooks, 1988). Rapid accumulation of algal solids, particularly in sand filters, make frequent back-flushing necessary in order to maintain through-flow. Use of automatic back-flushing (based on loss of head within the filter) make sand filters rather expensive, and unlikely to provide sufficient benefit to warrant current investigation.

Rock filters, by contrast, are a more "passive" technology which may have application to dairy farms. Rock filters may be physically similar to artificial gravel-bed wetlands (without plantings) or constructed as upflow filters. Rock filters have been used successfully to polish domestic sewage pond effluents (O'Brien, 1975; Swanson and Williamson, 1980) by removal of algal solids. Development of biofilms on the internal filter surfaces, capable of further nitrification and BOD conversion, is also likely to provide treatment benefits. The main problem with the use of these filters for dairy pond wastewaters is that they may eventually clog and, in the absence of any means of cleaning, need to be rebuilt. The slowly-degradable nature of grass fibre in dairy shed wastes is likely to result in rapid accumulation of organic solids, as noted for gravel-bed constructed wetlands not preceded by surface-flow wetlands (Tanner and Sukias, 1995).

Research need:
Removal of suspended solids from dairy shed pond effluents is desirable with existing pond design and will become even more important if pond design is modified to enhance algal growth and oxygenation of the wastewater. Further information on the settling characteristics, composition, and biodegradability of suspended solids in dairy pond discharges is required to evaluate the potential performance and sustainability of rock filter systems. This information will also be useful in modelling studies of pond systems and assessments of wetland and overland flow systems. If rock filters can be shown to have the potential to operate effectively over reasonable time-frames (>10 years) they should have application in tandem with other add-ons such as overland flow.

7.3.3 *Overland flow*

Overland-flow (Figure 3, option 9) is a well-tested technology for treatment of domestic wastewaters (eg. USEPA, 1984; WPCF, 1990), and recent studies have shown its potential for treatment of higher strength agricultural wastes such as piggery wastewaters after anaerobic lagoon pretreatment (Hubbard et al., 1994; Hawkins et al., 1995). In overland-flow systems wastewater is applied intermittently to grassed slopes (2-10% gradient). Wastewater essentially flows as a thin sheet across the soil surface, saturating the upper soil layers, and passing through the litter layer and grass sward. Suspended solids are removed by settling and filtration, and dissolved organic compounds and nutrients are sorbed by the soil and associated microbial biofilms. Plants also assimilate wastewater nutrients into their biomass. During the drying phase after wastewater application, aerobic degradation of organic matter and nitrification of ammonia occurs. Denitrification of accumulated nitrate may occur during subsequent flooded periods.

Improvement to dairy shed pond effluent quality might be simply achieved by permitting the effluent to flow overland in the riparian zone before reaching the receiving stream or drain. Where a suitable slope is available, the effluent could be intermittently spread over the designated land area (eg. 20 x 20 m^2 for an average-sized 180 cow system) through a perforated pipe before finally entering natural waterways. On steeper slopes or less permeable soils, the effluent would flow over the surface, with solids being trapped by pasture vegetation, and some of the nutrients being utilised for grass growth. On gentler slopes or more permeable soils, a proportion of the effluent will infiltrate the soil.

Overland flow treatment seems likely to conform to Maori spiritual beliefs that wastes must pass through land in order to be cleansed, and therefore be considered favourably by Regional Councils attempting to fulfil Treaty of Waitangi requirements. Regional Councils and farmers would need to be involved in research to demonstrate the practicality of overland flow systems as a form of treatment. Sampling overland flow effluent, as it reaches the waterway, is likely to be difficult, making routine monitoring of the final discharge problematic. If overland flow could be demonstrated to achieve its potential, it may be able to be designated a permitted activity, thereby avoiding the need for farmers to abandon their investment in ponds, and turn to alternatives such as spray-irrigation, which are management-intensive and may be unreliable.

The land used in overland-flow and riparian-zone systems would have to be fenced to prevent stock access while the soil was subject to waste applications. Regular cropping of grass is generally required to maintain sward vigour in normal overland flow set-ups, but direct grazing is unlikely to be a feasible means of vegetation management because of trampling effects on soil structure and the evenness of the soil surface. Such problems may be able to be overcome by the use of alternative plants (e.g. wetland species) which are able to provide a more sustainable high density sward under periodically saturated conditions without the need for grazing. In practice, despite being originally designed for subsurface-flow, most of the soil-based reed-bed wetland systems used in Europe operate predominantly via surface-flow of wastewaters through the litter and plant layer (Schierup et al., 1990; Hiley, 1995). High levels of treatment are still achieved in these essentially "overland-flow wetland" systems, which may provide a more practical option than "classic" overland-flow systems that rely on more management intensive grass swards. Enhancement of riparian vegetation could also be undertaken close to the waters edge to improve efficiency of treatment in this zone, particularly denitrification.

To avoid the need for manual or automated switching of flows between adjacent plots to achieve intermittent wastewater applications (generally 8 hours application/16 hrs rest), the second pond could be fluctuated (~5 cm) using a simple floating siphon control system or automated level-controlled pump to provide appropriate pulses of flow. In practice the pulsed discharges that result from daily shed wash-down may prove sufficient for simplified overland-flow systems.

Research need:
Of all the add-ons discussed here, overland flow would seem to have perhaps the greatest potential for application to existing dairy shed pond systems. We therefore regard overland flow systems as a priority for research. General design features and key processes operating in overland-flow treatment systems are reasonably well understood for domestic sewage treatment (e.g. USEPA, 1984; Kruzic and Schroeder, 1990; WPCF, 1990), although they have not been commonly applied in New Zealand. Identification and field testing of appropriate design and loading criteria are required for dairy waste stabilisation pond effluents, emphasising simplicity and reliability of design and operation. Work should be done on overland flow in tandem with other enhancements, including shallow-depth ponds (expected to have high algal biomasses) and simple wetlands or rock filters which can potentially remove the algal solids. Studies of the discharge flow patterns of dairy shed ponds will be useful for evaluating the need to specifically create intermittent flow regimes, otherwise simple reliable devices will need to be developed and tested to provide pulsed flows.

7.3.4 *Constructed wetlands*

NIWA has undertaken considerable research on the use of constructed wetlands (Figure 3, option 10) for treatment of dairy and piggery wastewaters. Basic principles of constructed wetland design and treatment expectations for dairy shed wastewaters have been summarised in Cooke et al. (1992). The performance results from farm-scale wetland trials (Sukias and Tanner, 1993; Sukias and Tanner, 1995; Sukias and Tanner, 1996) have generally been less favourable than expected based on initial pilot-scale trials carried out on the Ruakura Research Farm of the Dairying Research Corporation (Tanner et al., 1995a; 1995b). However the trials show that constructed wetlands incorporating initial surface-flow channels and final gravel zones can effectively reduce BOD and SS levels in dairy pond discharges. Although total N removal can be substantial, ammonia reduction is generally between 10 and 50% at realistic loading rates. Significantly higher N removal is only likely to be practical if pond pre-treatment performance can be substantially enhanced, by for instance using mechanical aeration to promote initial nitrification of the wastewater (e.g. van Oostrom and Russell, 1994). Phosphorus removal via sediment adsorption is generally low over the longer term, unless specific P adsorbing substrata are used (e.g. allophanic clays, iron rich minerals such as melter slag, calcium-rich fly ashes).

Guidelines for constructed wetland treatment of dairy shed pond discharges are presently in preparation, with completion planned for late 1996. Three basic wetland treatment levels are envisaged, with highest levels of treatment performance requiring initial treatment in a partially aerated pond system. Ongoing FRST-funded research being carried out by NIWA is investigating the sustainability of treatment performance and the fate of nutrients and organic matter in mature wetland systems, receiving dairy pond effluents. Generic wetland treatment research is investigating the use of water level fluctuations to enhance ammonia removal and evaluating treatment processes along the length of wetlands over a range of wastewater pretreatment levels.

Research need:
Further studies of constructed wetland treatment of dairy pond effluents should focus on their performance treating discharges from aerated and other upgraded pond systems. Being predominantly anaerobic environments, constructed wetlands are likely to provide valuable solids and N removal (via denitrification), as well as buffering receiving streams from peaks in effluent loading. Organic carbon inputs from plant growth in these wetlands are likely to promote denitrification in wetland systems.

7.3.5 *Rotating biological contactors*

Rotating biological contactors (RBCs) (Figure 3, option 11) comprise a series of semi-submerged, large diameter discs mounted on a rotating shaft within a tank. Essentially a variation of fixed film reactors, the discs rotate slowly through the wastewater lifting a thin film of wastewater out of the tank and allowing it to slowly percolate back down over the attached biomass. The movement of the discs shears off excess film biomass resulting in a largely self-maintaining treatment process. Cooke et al. (1992) summarised the key practical and performance attributes of these systems, and considered RBCs to be the most suitable "high-tech" option among those that they reviewed for high-ammonia effluents such as dairyshed wastewaters. Results of laboratory-scale RBC experiments carried out at NIWA, have shown that over 90% removal of ammonia-N from dairy pond wastewaters (initial concentration 60 g m^{-3}) is possible at residence times of <1 day.

Research need:
We recommend that pilot studies be carried out in association with existing manufacturers of RBC systems, to evaluate their performance treating dairy pond effluents, and their practical and economic attributes for farm use.

7.3.6 *Land irrigation*

Irrigation of dairy pond effluents onto pasture land (Figure 3, option 12) has the advantage of conforming closely to Maori values regarding waste disposal. If the waste is spread over adequate area to ensure its safe assimilation, and the pond system is set up to provide storage during wet (non-sprayable) periods, then such a system would probably be deemed to be a permitted activity under RMA rules being developed by most Regional Councils. Based on N-content, effluents treated in 2-pond systems are likely to require irrigation areas of around a third of that required for raw dairy wash-down water and 45% of that required for anaerobically treated effluents. Factors other than N loading may however constrain the application rate of 2-pond effluents to levels below this, particularly hydraulic loading.

Present land application of dairy wastes stored in anaerobic ponds (generally including resuspended sludge accumulations) is carried out about three times per year by contractors using specialised machinery. The reduced land area requirements (or increased areal application rates) for irrigation of

2-pond treated effluents may make land application more manageable for farmers to do themselves, as it will require less frequent moving of irrigators and be able to be done using simpler spray equipment. It has the disadvantages of recycling somewhat less nutrients (although much of the nutrients "lost" from the effluent would eventually be returned to the land during pond desludging operations) and requiring the provision and maintenance of both pond and irrigation systems.

The irrigation area requirements for a 200 cow herd are likely to be in the order of 2.5 ha (cf. ~8 ha for stored anaerobic pond slurry) assuming a maximum N application rate of 150 kg ha^{-1}. Additional land areas would be required for periodic spreading of pond sludges. This could be done on land less amenable to year-round spray irrigation. Effluent applications would have to be made more frequently than for more concentrated wastewaters so as to not overload the soil's hydraulic capacity to effectively absorb and assimilate the applied wastes. Lower levels of organic C in effluents from 2-pond systems may reduce potential N losses via denitrification.

Research need:
The practical advantages and disadvantages for farmers of the various land application options (particularly direct application versus applying from existing anaerobic or facultative ponds) should be investigated. In addition, existing criteria for land application should be reviewed in the light of these different potential options to ensure groundwater is protected from nitrate contamination.

8. SUMMARY AND CONCLUSIONS

Dairy shed wastewaters in New Zealand are commonly treated by lagoon systems consisting of an anaerobic pond followed by a facultative pond. These dairy shed pond systems, originally developed in the 1970's, have resulted in major reductions in point-source loadings of BOD and suspended solids to streams and rivers. However, recently ponds have come to be regarded less favourably for treatment of dairy shed wastewaters, because high dilutions are still required to safely assimilate pond effluents in receiving streams.

This review discusses treatment processes within stabilisation ponds generally, and then examines dairy pond systems in the light of this conceptual understanding. To define the treatment standards necessary for dairy shed waste discharges, the quality of current dairy shed stabilisation pond effluents is evaluated from a receiving water perspective. The "priority pollutants" in dairy pond effluent, ranked according to the required dilution factors to meet receiving water guidelines, are: *nutrients* promoting nuisance growths, *faecal indicator bacteria* that measure potential risk to human and livestock health, *ammoniacal nitrogen* which can be toxic to stream life and also contributes significantly to oxygen depletion, and *suspended solids* for their effects on stream life and aesthetic quality.

Anaerobic dairy ponds generally work well at their primary function of removing suspended solids and BOD (mainly by sedimentation), although some simple measures to reduce sludge carryover to the second pond are recommended. In contrast, facultative dairy ponds appear to provide little additional treatment of dairy shed wastewaters. The restricted treatment is attributed to inadequate oxygen supply due to low algal abundance. Restricted light penetration into the wastewaters, which are strongly coloured by dissolved humic compounds, and ammonia toxicity, probably account for the low algal abundance and consequent limited oxygenation.

A range of methods for improving facultative (and anaerobic) dairy pond performance and for providing further treatment are summarised and evaluated. These approaches summarised in Figure 3, fall into a natural hierarchy, from pond modifications (e.g. additional pond area and shallow depth ponds) to simple "add-ins" to existing pond facilities (e.g. mechanical aeration), to supplementary "add-ons" (e.g. maturation ponds, constructed wetlands and overland flow systems).

In most instances, research or demonstration by manufacturers of these modifications is required to assess their applicability to dairy shed wastewaters and define their treatment capabilities under New Zealand farming conditions.

Mechanical aeration is an "add-in" option which shows high potential to improve the performance of dairy ponds. Research by NIWA, funded by DRI, is currently underway on aeration, and early results appear promising. Use of intermittent aeration is likely to promote increased N removal via denitrification, as well as substantially reducing energy costs. Combining geotextile support surfaces and "add-ons" such as constructed wetlands with aerated ponds should provide additional treatment enhancement.

"Add-ons" to pond systems, by their very nature, provide buffering of the inherent fluctuations in pond effluent quality, as well as providing additional treatment. "Add-ons" can be designed to remove pollutants for which the existing ponds are not designed. For example, constructed wetlands and overland-flow provide nutrient removal, particularly nitrogen, as well as supplementary BOD, SS and faecal coliform removal. Maturation ponds should provide very effective natural disinfection. Information from pilot- and farm-scale trials of constructed wetland carried over the last few years are presently being incorporated into a guideline document. Further studies are needed of wetland treatment of aerated pond effluents, high in nitrate.

Currently NIWA is investigating relationships between algal abundance, light penetration and other key physico-chemical variables, and treatment performance in six dairy farm pond systems. The aim of this study is to identify the factors causing the relatively poor performance of current dairy ponds, and provide basic information for development of improved pond systems. There is also an urgent need for reliable information on seasonal and diurnal fluctuations in pond discharge flows. This information is needed to (a) improve predictions of the impacts of current discharges on receiving waters, (b) assist the development of appropriate treatment targets for dairy shed wastewaters, and (c) design effective "add-on" systems to supplement pond treatment.

Key future research directions identified include evaluating the potential of: (a) shallow pond systems (which are expected to develop high algal biomasses and a better-oxygenated wastewater), (b) additional facultative and/or maturation ponds (to provide an extra level of treatment, especially disinfection, and buffer against poor effluent quality excursions), (c) simplified overland flow, constructed wetlands and rock-filter systems (to remove algal solids and buffer receiving waters from pond effluent), and (d) rotating biological contactors (to enhance biological oxidation of organic compounds and ammonia).

Table 1: Key contaminants in dairy oxidation pond effluents and predicted dilution factors required to safeguard water quality and aquatic life.

Variable	Effluent concentration[4]		Receiving water guideline[5]	Dilution factor required[6]		To protect
	median	95 percentile		median	95 percentile	
BOD[1]	98	241	5	>20	>50	Aquatic life[7]
Suspended solids	198	804	4	>50	>201*	Aquatic life[8]
Ammoniacal-N	75	191	0.76	>99	>248	Aquatic life–Salmonoids[9]
			0.22	>341	>868	Aquatic life–NZ fauna[10]
BOD + NBOD[2]	413	1068	5	>83	>214	Aquatic life[7,11]
Faecal coliforms[3]	70, 000	540,000	1000	>70	>540	Crop irrigation & stock drinking[12].
			200	>350	>2700	Contact recreation[13]
Dissolved inorganic N	75	216	0.04–0.1	>750–1875	>2160–5400*	Avoid undesirable algal growths[14]
Dissolved reactive P	12.2	17.1	0.015–0.03	>407–813	>570–1140*	Avoid undesirable algal growths[14]

1 Biochemical oxygen demand; a measure of the microbial oxygen consuming capacity of organic compounds in wastewater.
2 Nitrogenous BOD; a measure of potential oxygen consumption due to microbial oxidation of ammonium to nitrate.
3 Indicator of pathogenic bacterial levels
4 Data from Hickey et al. (1989a); all Concentration units g m^{-3}; except faecal coliforms, number cfu 100 mls^{-1}
5 Suggested maximum values to protect receiving waters under the RMA, 1991. See notes 7–14.
6 Minimum dilution required assuming no background contamination.
7 To avoid serious deoxygenation effects on fish and other biota (USEPA, 1976). USEPA (1986) suggests that this level is likley to result in moderate production impairment of salmonids (excluding more sensitive embryo and larval stages) and slight to moderate impairment of non-salmonid fish species.
8 To avoid adverse impacts on benthic invertebrate communities (Quinn and Hickey 1994).
9 To avoid ammonia toxicity to fish and invertebrates; salmonoid criteria, 4 day, pH 8, 20°C (USEPA, 1986)
10 Provisional guideline to avoid ammonia toxicity to NZ native invertebrate species, based on 4 day, pH 8, 20°C USEPA (1986) value multiplied by the ratio of the "final acute value" (FAV) for toxicity data obtained by Hickey & Vickers (1994; = 0.15 g m^{-3}) to the FAV derived by the USEPA (1986; = 0.52 g m^{-3})
11 Total potential oxygen demand calculated as the sum of the BOD and NBOD (see Cooper 1986, Hickey et al. 1989a).
12 Faecal coliforms: Guideline for safe irrigation of crops eaten uncooked (WHO, 1989) and livestock drinking water (ANZEC, 1992)
13 Faecal coliforms: Guideline for human bathing and contact recreation (DoH, 1992).
14 Maximum nutrient concentrations to avoid undesirable algal proliferations (MfE, 1992).
* Average conditions, rather than short-term fluctuations likely to be of most relavence when considering impacts of SS and nutrient enrichment.

Table 2. Options for upgrading of dairy shed ponds

Anaerobic ponds

Upgrade option	*Rationale*
Design to facilitate desludging.	Ensure sludge removal is complete.
Baffle-board or double "tee".	Minimise entrainment of gas-buoyed solids in effluent.

Facultative ponds ("Add-ins")

Upgrade option	*Rationale*
Inlet/outlet positioning.	To minimise short circuiting of wastewater.
Outlet structures.	Designed to minimise algal entrainment.
Shading of outflow region.	Reduce algal entrainment, and thereby algal BOD and SS in pond effluent.
Shallow depth ponds.	Shallower ponds permit better light penetration, with consequently better algal growth and oxygen supply.
Desludging.	Ensure that sediment oxygen demand is minimised.
Mechanical aeration.	Supply sufficient oxygen for good BOD removal and oxidation of ammoniacal-N to nitrate. Odour control.
Aeration with biofilm support surfaces.	Encourage biofilm formation to enhance BOD removal and nitrification.
Baffling or barriers.	Induces plug-flow (or multiple pond cascade) giving improved removal of SS, BOD and (particularly) bacteria.

Further treatment of pond effluent ("Add-ons")

Upgrade option	*Rationale*
Maturation ponds.	Further removal of BOD and (particularly) faecal bacteria and pathogens. Buffering against poor effluent quality "excursions". Nitrification.
Rock filters.	Reduce algal or other solids in pond effluent. Distribute effluent flow for overland flow treatment.
Overland flow.	Buffering of receiving waters. Further removal of contaminants, including algal solids.
Wetlands.	Buffering of receiving waters. Further removal of contaminants.
Increased pond size.	Improve treatment to account for increasing herd size or under-estimation of waste production.

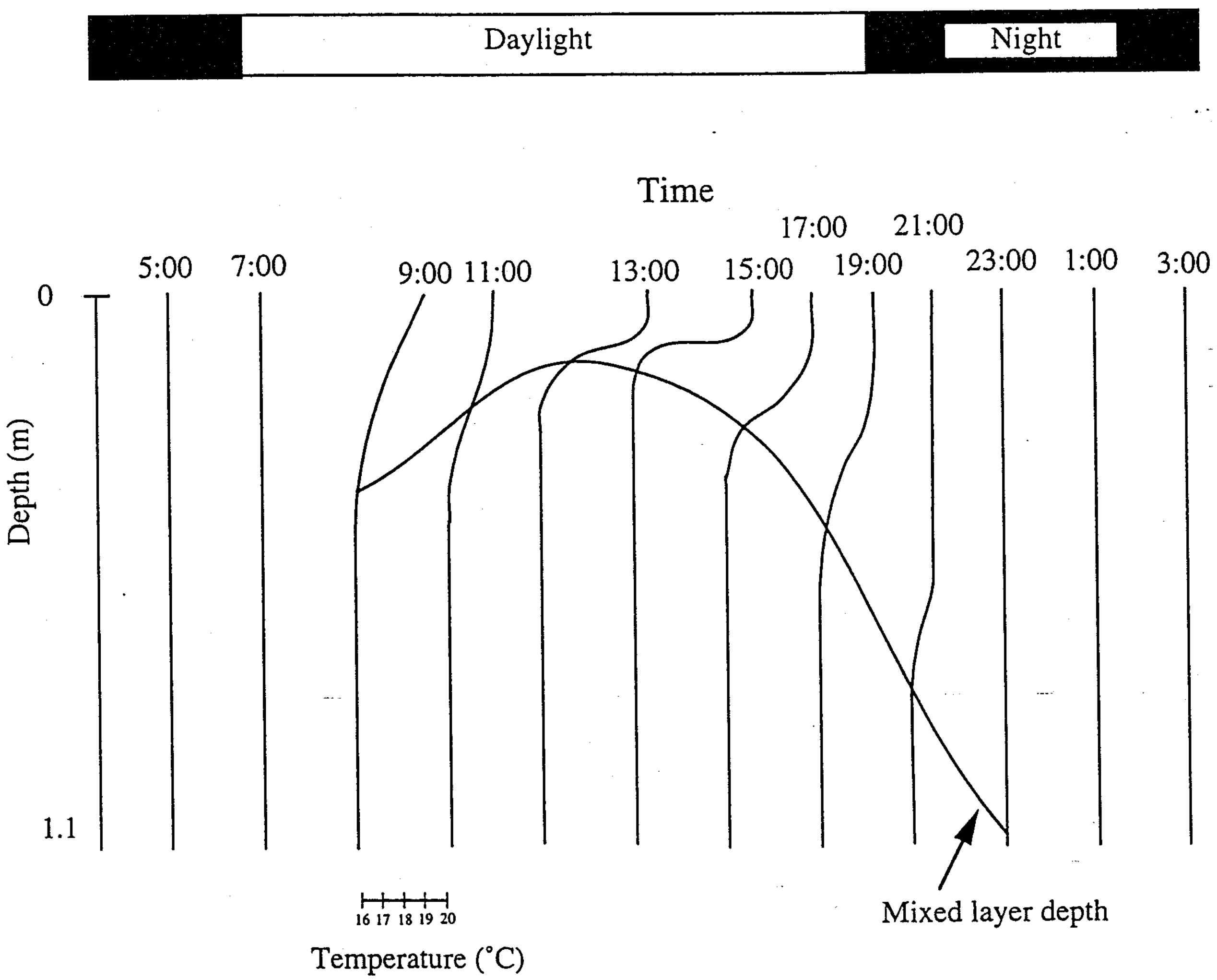


Figure 1. Schematic showing the time course of thermal stratification of a facultative dairy shed pond on a sunny day. Warming of the surface water by sunlight during the morning leads to formation of a very shallow warm layer (epilimnion) which tends to be mixed down by wind action later in the day (notice the "sharpening" of the base of the epilimnion). After about mid-afternoon, surface cooling begins when the insolation declines, and the epilimnion deepens and its temperature contrast with the hypolimnion reduces. This process continues through the evening aided by convective sinking of dense parcels of water cooled at the water surface. Eventually the epilimnion cools to the same temperature as the hypolimnion and overturn occurs with mixing of the waters in the previously seperate layers.

Each time trace has been off-set from the previous trace. The temperature scale should be read with the same off-set, ie. at 1.1 m the temperature is 16 °C for each trace.

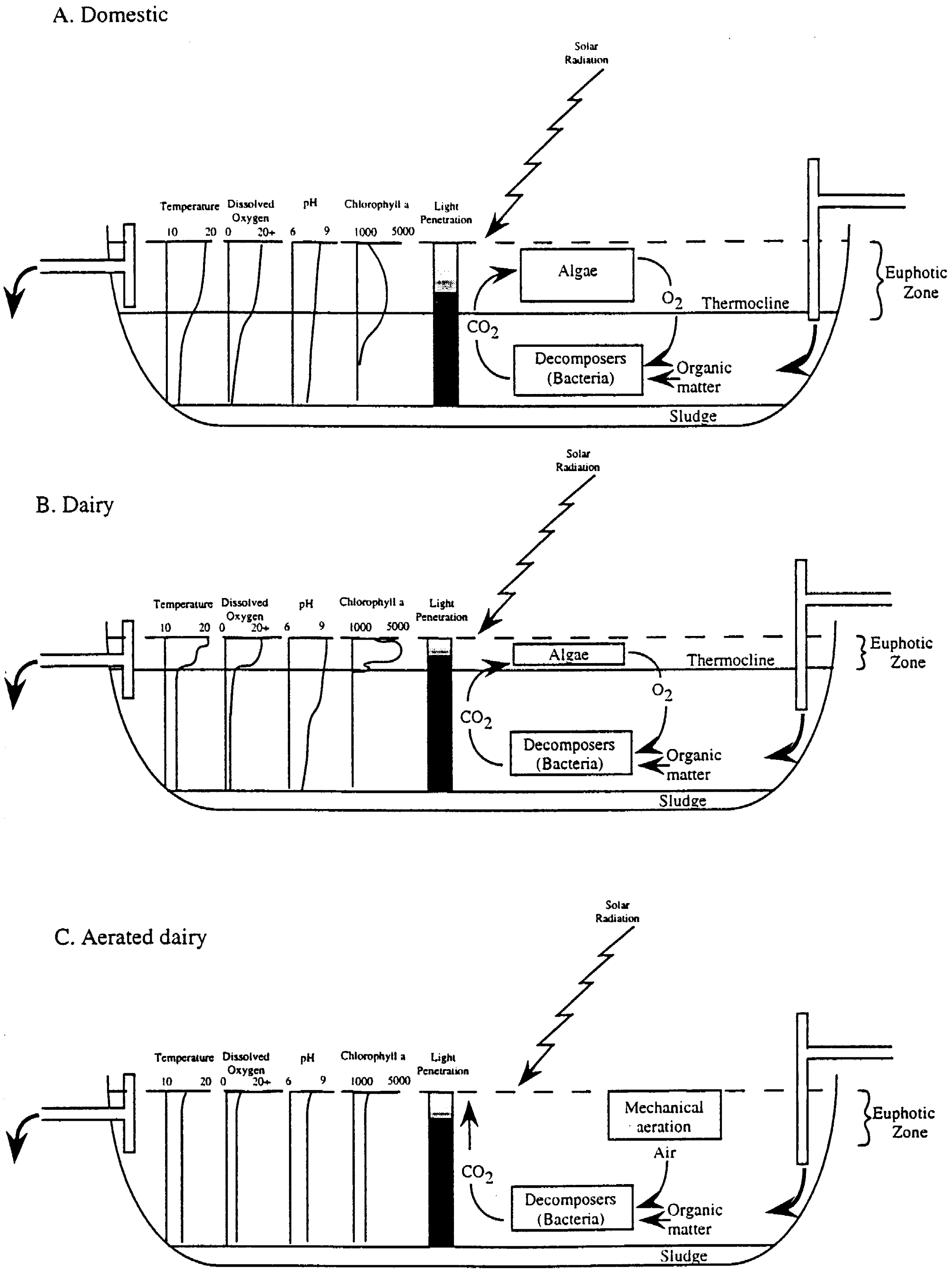


Figure 2.
Styalised representation of facultative waste stabilisation ponds. A. Domestic sewage pond showing daytime stratification. B. Facultative dairy shed pond showing daytime stratification. C. Aerated facultative dairy pond, where algal photosynthesis is limited, and stratification does not occur.

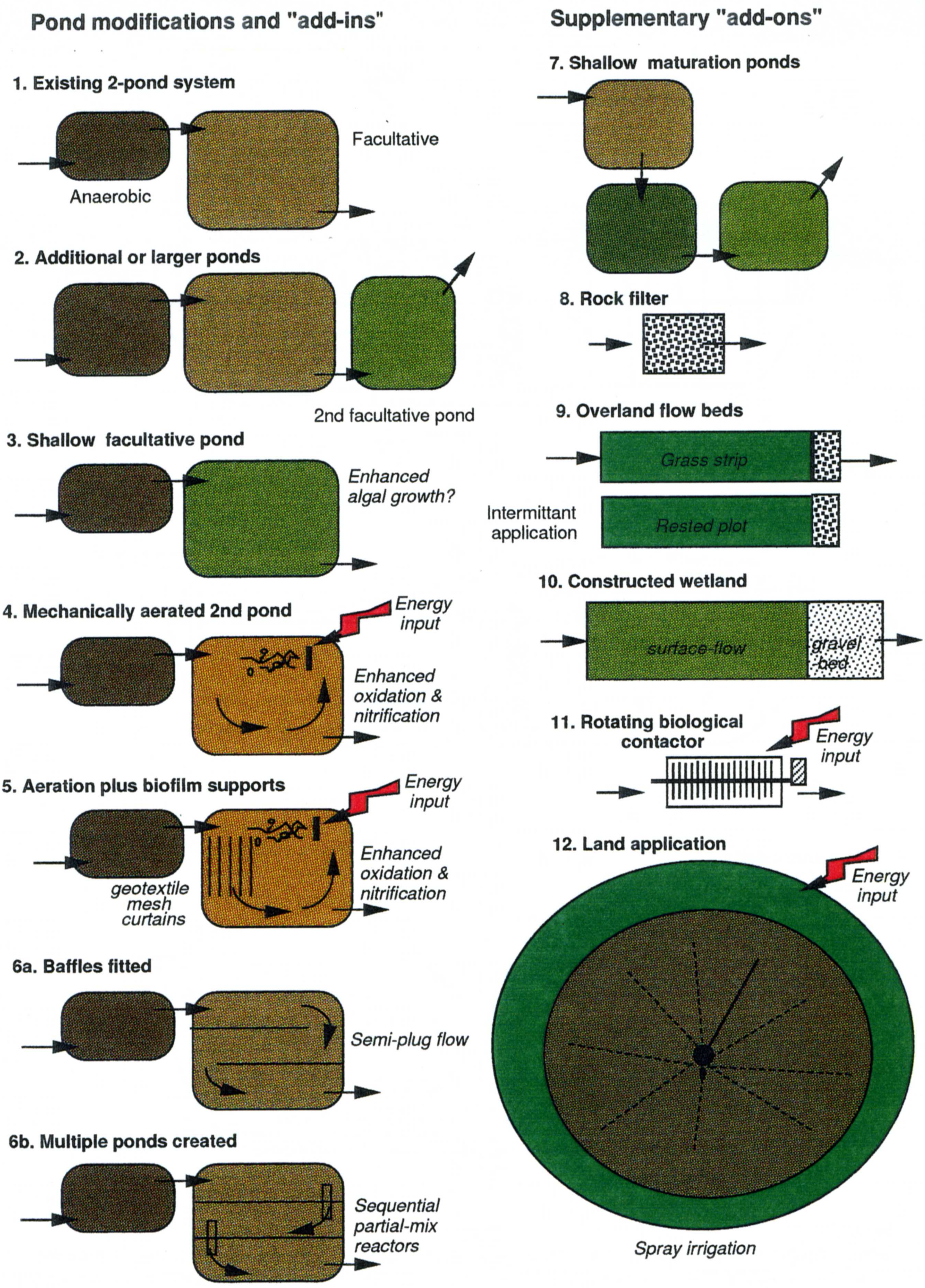


Figure 3. Schematic summary of modifications, "add-ins" and "add-ons" to existing 2-stage dairy shed wastewater treatment ponds

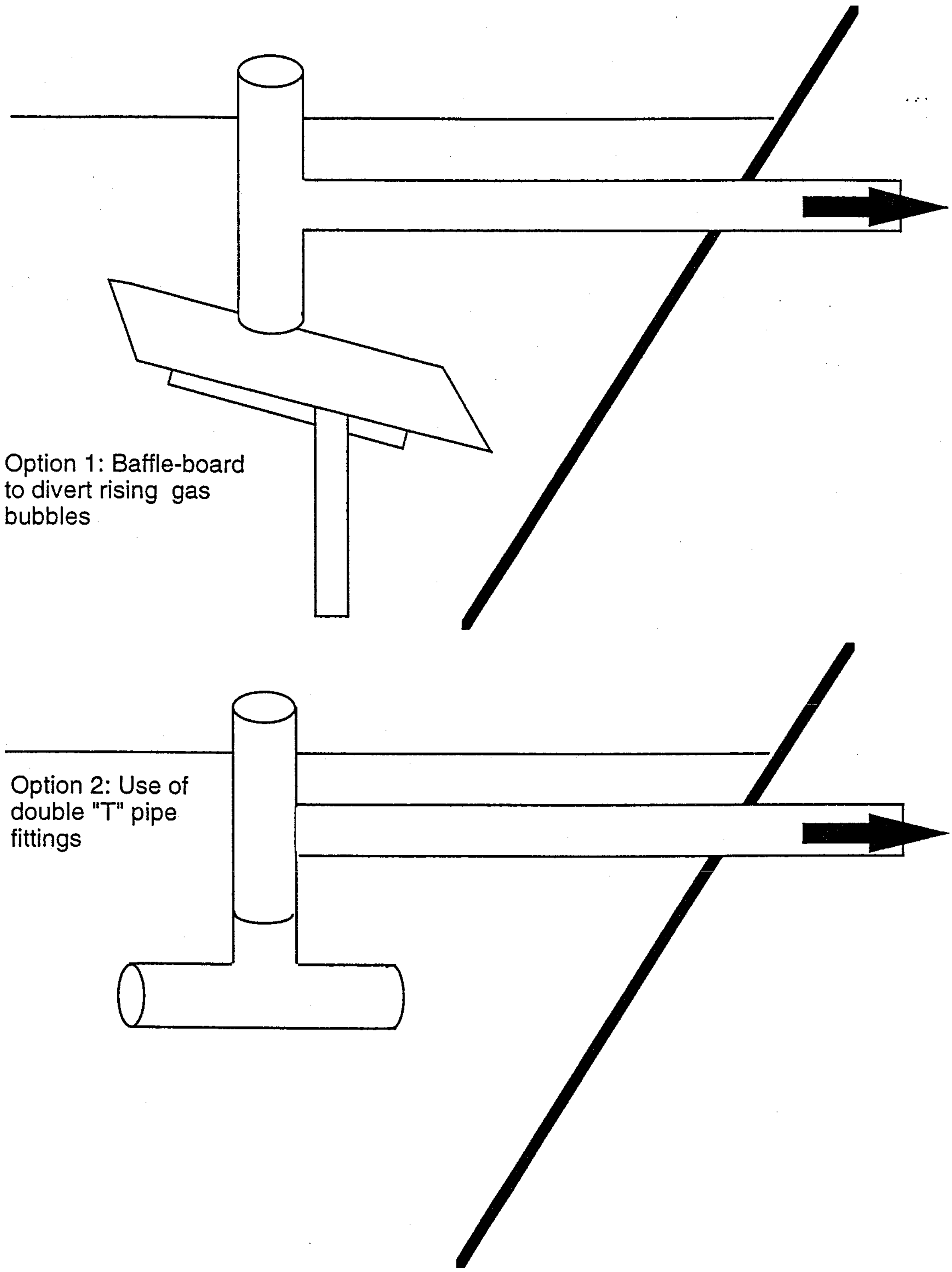


Figure 4: Anaerobic pond outlet pipe modifications to reduce entrainment of rising gas bubbles and associated sludge.

9. GLOSSARY

aerobic	requiring free or dissolved oxygen.
allophanic	of allophane clays (containing hydrous aluminium silicate) with high P sorption potential
anaerobic	requiring reduced conditions.
anoxic	deficient in oxygen.
benthic	occurring in substrate at bottom of pond (*or lake or sea*).
biofilm	thin layer of living material containing bacteria, fungi and other micro-organisms living on solid substrate.
BOD	biochemical oxygen demand, a measure of the amount of oxygen required to neutralise organic wastes by microbial processes.
decarboxylation	(of acetic acid) removal of carboxyl acid group –COOH
denitrification	microbial conversion of nitrate to N_2O or N_2 gasesin anaerobic conditions.
epilimnion	upper layer of water in stratified water bodies.
euphotic	zone of water where light penetration is sufficient to allow photosynthesis to occur (*defined as depth to which 1% of incident light can penetrate*).
facultative	(*in this context*) containing both aerobic and anaerobic metabolisms.
heterotrophic	requiring pre-existing organic material as a source of nutrient.
hypolimnion	lower layer of water in stratified lake (*or pond*).
maturation ponds	sequence of shallow ponds designed to facilitate photoinactivation of pathogens.
methanogenic	methane synthesysing (bacteria)
nitrification	conversion of ammonia into nitrate.
oxic	containing oxygen.
phaeophytin	breakdown product of chlorophyll
photochemical inactivation	inactivation of pathogens due to the presence of certain chemical compounds formed by sunlight.
photosynthesis	process in which energy of sunlight is used by green plants to build up complex substances from carbon dioxide and water, with subsequent release of oxygen.
phototrophic	organisms using sunlight as their energy source, e.g. algae.
phytoplankton	microscopic free floating algae.
stratification	seperation into layers.
supersaturation	above equilibrium concentration (>100% saturation).
symbiotic	co-operative association of two different organisms.
thermocline	point of thermal stratification,
volatilisation	conversion from liquid to a vapour (evaporation).

10. REFERENCES

Abeliovich, A. and Azov, Y. (1976). Toxicity of ammonia to algae in sewage oxidation ponds. *Applied and Environmental Microbiology* **31**(6): 801-806.

Abeliovich, A. and Weisman, D. (1978). Role of heterotrophic nutrition in growth of the alga *Scenedesmus obliquus* in high-rate oxidation ponds. *Applied and Environmental microbiology* **35**(1): 32-37.

Agunwamba, J. C. (1993). Desludging interval in a waste stabilization pond: a Markov Approach. *Journal of Environmental Management* **38**: 289-299.

ANZECC (1992). Australian Water Quality Guidelines. Australian and New Zealand Environment and Conservation Council, . Canberra.

Araki, H., Koga, K., Inome, K., Kusuda, T. and Awaya, Y. (1990). Intermittent aeration for nitrogen removal in small oxidation ditches. *Water Science and Technology* **22**(3/4): 131-138.

ASAE (1990). Design of anaerobic lagoons for animal waste management. American Society of Agricultural Engineers, Engineering Practice ASAE EP403.1.

ASAE (1991). Design of anaerobic lagoons for animal waste management. ASAE Engineering Practice: ASAE EP403.1. American Society of Agricultural Engineers.

Azov, Y. and Goldman, J. C. (1982). Free ammonia inhibition of algal photosynthesis in intensive cultures. *Applied and Environmental Microbiology* **43**(4): 735-739.

Azov, Y., Shelef, G. and Moraine, R. (1982). Carbon limitation of biomass production in high-rate oxidation ponds. *Biotechnology and Bioengineering* **24**: 579-594.

Barth, C. L. and Kroes, J. (1985). *Livestock waste lagoon sludge characterization*. Proceedings of the Agricultural Waste Utilization and Management, Chicago, Il, American Society of Agricultural Engineers.

Baskaran, K., Scott, P. H. and Conner, M. A. (1992). Biofilms as an aid to nitrogen removal in sewage treatment lagoons. *Water Science and Technology* **26**(7-8): 1707-1716.

Calderon, R. L., Mood, E. W. and DuFour, A. P. (1991). Health effects of swimmers and nonpoint sources of contamination. *International Journal of Environmental Health Research* **1**: 21-31.

Caraco, N. F., Cole, J. J. and Likens, G. E. (1989). Evidence for sulphate-controlled phosphorus release from sediments of aquatic systems. *Nature* **341**: 316-318.

CCREM (1987). Canadian Water Quality Guidelines. Task Force on Water Quality Guidelines, Canadian Council of Resource and Environment Ministers, . Ottawa.

Cohen, J. and Shuval, H. I. (1972). Coliforms, fecal coliforms, and fecal streptococci as indicators of water pollution. *Water, Air, and Soil Pollution* **2**: 85-95.

Constable, J. D., Scott, P. H. and Conner, M. A. (1989). *Fixed bed nitrification as a potential means of enhancing nitrogen removal rates in a sewage lagoon*. Proceedings of the Australian Water and Wastewater Association, Canberra.

Cooke, J. G., Hickey, C. W. and Tanner, C. C. (1992). Critical review of techniques for the reduction of ammonium in rural point source discharges. Environmental and Planning Division, Auckland Regional Council, Technical Publication 8. Auckland, NZ.

Cooper, A. B. (1986). Developing management guidelines for river nitrogenous oxygen demand. *Journal of the Water Pollution Control Federation* **58**: 845-852.

Curtis, T. P., Mara, D. D., Dixo, N. G. H. and Silva, S. A. (1994). Light penetration in waste stabilization ponds. *Water Research* **28**(5): 1031-1038.

Curtis, T. P., Mara, D. D. and Silva, S. A. (1992). Influence of pH, oxygen, and humic substances on ability of sunlight to damage fecal coliforms in waste stabilization pond water. *Applied and Environmental Microbiology* **58**(4): 1335-1343.

Dakers, A. J. and Painter, D. J. (1983). *Livestock waste management in New Zealand.* Proceedings of the 15th New Zealand biotechnology conference, Massey University, Palmerston North, New Zealand.

Davies-Colley, R. J. (1995). *Effluent quality concerns with sewage lagoons in New Zealand.* Proceedings of the New Zealand Water and Wastes Association Annual Conference, Ellerslie Conference Centre, Auckland.

Davies-Colley, R. J., Hickey, C. W. and Quinn, J. M. (1995). Organic matter, nutrients, and optical characteristics of sewage lagoon effluents. *New Zealand Journal of Marine and Freshwater Research* **29**: 235-250.

Davies-Colley, R. J., Vant, W. N. and Smith, D. G. (1993). *Colour and clarity of natural waters. Science and management of optical water quality*. Ellis Horwood, London.

DEC (in prep.). Dairying and the environment– managing farm dairy effluent. Dairying and the Environment Committee of the New Zealand Dairy Board, Draft manual.

DoH (1992). Provisional Microbiological Water Quality Guidelines for Recreational and Shellfish-Gathering Waters in New Zealand. Public Health Services, Department of Health, McBride, G. B., Cooper, A. B. and Till, D. G. Wellington, NZ.

Ellis, K. V. (1983). Stabilization ponds: Design and operation. *CRC critical reviews in environmental control* **13**(2): 69-102.

Environment Waikato (1993). Dairy shed waste treatment systems. Environment Waikato, . Hamilton.

Fallowfield, H. J. and Garrett, M. K. (1985). The treatment of wastes by algal culture. *Journal of Applied Bacteriology Symposium Supplement* : 187S–205S.

Ferrara, R. A. and Avci, C. B. (1982). Nitrogen dynamics in waste stabilization ponds. *Journal of the Water Pollution Control Federation* **54**(4): 361-369.

Fujioka, R. S. and Narikawa, O. T. (1982). Effect of sunlight on enumeration of indicator bacteria under field conditions. *Applied and Environmental Microbiology* **44**(2): 395-401.

Ghrabi, A. and Ferchichi, M. (1994). Sediment accumulation in a series of four pilot-scale stabilization ponds. *Water Science and Technology* **30**(8): 281-284.

Gloyna, E. F. and Tischler, L. F. (1981). Recommendations for regulatory modifications: the use of waste stabilization pond systems. *Journal of the Water Pollution Control Federation* **53**(11): 1559-1563.

Goldman, J. C., Azov, Y., Riley, C. B. and Dennett, M. R. (1982). The effect of pH in intensive microalgal cultures. I. Biomass regulation. *Journal of Experimental Marine Biology and Ecology* **57**: 1-13.

Goldman, J. C., Azov, Y., Riley, C. B. and Dennett, M. R. (1982). The effect of pH in intensive microalgal cultures. I. Species competition. *Journal of Experimental Marine Biology and Ecology* **57**: 15-24.

Gu, R. and Stefan, H. G. (1995). Stratification dynamics in wastewater stabilization ponds. *Water Research* **29**(8): 1909-1923.

Hanes, N. B., Sarles, W. B. and Rolich, G. A. (1964). Dissolved oxygen and survival of coliform organisms and enterococci. *Journal of the American Water Works Association* **56**: 441-446.

Hart, B. T. (1974). A Compilation of Australian Water Quality Criteria. Australian Water Resources Council, Technical Paper No. 7. Canberra.

Hawkes, H. A. (1983). Stabilization ponds. *Ecological aspects of used-water treatment*. C. R. Cards Ed. Academic Press, London. 163–217.

Hawkins, G. L., Hill, D. T., Rochester, E. W. and Wood, C. W. (1995). Evaluation of overland flow treatment for swine lagoon effluent. *Transactions of the ASAE* **38**(2): 397-402.

Hickey, C. W. (1988). Benthic chamber for use in rivers: testing against oxygen mass balances. *Journal of Environmental Engineering, ASAE* **114**: 828-845.

Hickey, C. W., Quinn, J. M. and Davies-Colley, R. J. (1989a). Effluent characteristics of dairy shed oxidation ponds and their potential impacts on rivers. *New Zealand Journal of Marine and Freshwater Research* **23**: 569-584.

Hickey, C. W., Quinn, J. M. and Davies-Colley, R. J. (1989b). Effluent characteristics of domestic sewage oxidation ponds and their potential impacts on rivers. *New Zealand Journal of Marine and Freshwater Research* **23**: 585-600.

Hickey, C. W. and Rutherford, J. C. (1986). Agricultural point source discharges and their effects on rivers. *New Zealand Agricultural Science* **20**: 104-110.

Hickey, C. W. and Vickers, M. L. (1994). Toxicity of ammonia to nine native New Zealand freshwater invertebrate species. *Archives of environmental contamination and toxicology* **26**: 292–298.

Hiley, P. D. (1995). The reality of sewage treatment using wetlands. *Water Science and Technology* **32**(3): 329-338.

Houng, H. J. S. and Gloyna, E. F. (1984). Phosphorous models for waste stabilization ponds. *Journal of Environmental Engineering* **110**(3): 550-561.

Hubbard, R. K., Newton, G. L., Davis, J. G., Dove, R., Lowrance, R. and Vellidis, G. (1994). *Grass-riparian zone buffer systems for filtering swine lagoon waste*. Proceedings of the Constructed Wetlands for Animal Waste Management, Lafayette, Indiana, Department of Forestry and Natural Resources, Purdue University.

Ip, S. Y., Bridger, J. S., Chin, C. T., Martin, W. R. B. and Raper, W. G. C. (1982). Algal growth in primary settled sewage: The effects of five key variables. *Water Research* **16**: 621-632.

James, A. (1987). An alternative approach to the design of waste stabilization ponds. *Water Science and Technology* **19**(12): 213-218.

Juanico, M. (1991). Should waste stabilization ponds be designed for perfect-mixing or plug-flow? *Water Science and Technology* **23**: 1495-1502.

Kilani, J. S. and Ogunrombi, J. A. (1984). Effect of baffles on the performance of model waste stabilization ponds. *Water Research* **18**(8): 941-944.

King, D. L. (1976). Changes in water chemistry induced by algae. *Ponds as a Wastewater Treatment Alternative* . E. F. Gloyna, J. F. Malina Jr. and E. M. Davis Eds. University of Texas, Austin TX. 73-84.

Koottatep, S., Leesanga, C. and Araki, H. (1993). Intermittent aeration for nitrogen removal in small aerated lagoon. *Water Science and Technology* **28**(10): 335-341.

Kruzic, A. P. and Schroeder, E. D. (1990). Nitrogen removal in the overland flow treatment process—removal mechanisms. *Research Journal of the Water Pollution Control Federation* **62**(7): 867-875.

Lallai, A., Mura, G., Miliddi, R. and Mastinu, C. (1988). Effect of pH on growth of mixed cultures in batch reactor. *Biotechnology and Bioengineering* **31**: 130-134.

MAF (1975). Effluent Ponds: Construction recommendations. Ministry of Agriculture and Fisheries, Aglink .

MAF (1980). Effluent Disposal: From Wintering Sheds, on dairy farms. Ministry of Agriculture and Fisheries, Aglink FPP 290.

MAF (1985). Effluent Ponds: Construction recommendations. Ministry of Agriculture and Fisheries, Aglink FPP 291.

MAF Policy (1994). Dairy shed wastewater treatment ponds. Ministry of Agriculture and Fisheries, Agriculture Policy, Technical Paper 94/17.

Mara, D. D. (1987). Waste stabilisation ponds: problems and controversies. *Water Quality International* **1**: 20-22.

Mara, D. D., Alabaster, G. P., Pearson, H. W. and Mills, S. W. (1992). *Waste stabilisation ponds: A design manual for eastern Africa*. Lagoon Technology International Ltd, Leeds, England.

Mara, D. D. and Marecos do Monte, M. H. F. (1990). The design and operation of waste stabilization ponds in tourist areas of Mediterranean Europe. *Water Science and Technology* **22**(3/4): 73-76.

Mara, D. D. and Mills, S., W (1994). Who's afraid of anaerobic ponds? *Water Quality International* **2**: 34-36.

Marais, G. v. R. (1974). Faecal bacterial kinetics in stabilization ponds. *Journal of the Environmental Engineering Division, ASAE* **100**: 119-139.

Mason, I. G. (1994). *Oxygen demand removal in a facultative waste stabilisation pond treating dairy shed wastewater*. Proceedings of the Engineering in Agriculture, Lincoln University, Christchurch, New Zealand.

Mayo, A. W. (1995). Modelling coliform mortality in waste stabilization ponds. *Journal of Environmental Engineering* **121**(2): 140-152.

Mayo, A. W. and Noike, T. (1994). Response of mixed cultures of *Chlorella vulgaris* and heterotrophic bacteria to variation of pH. *Water Science and Technology* **30**(8): 285-294.

McBride, G. B. (1993). Comment. *International Journal of Environmental Health Research* **3**: 115-116.

MfE (1992). Guidelines for the control of undesirable biological growths. Ministry for the Environment, Water Quality Guideline no. 1. Wellington.

Middlebrooks, E. J. (1988). Review of rock filters for the upgrade of lagoon effluents. *Journal of the Water Pollution Control Federation* **60**(9): 1657-1662.

Middlebrooks, E. J., Middlebrooks, C. H., Reynolds, J. H., Watters, G. Z., Reed, S. C. and George, D. B. (1982). *Wastewater Stabilization Lagoon Design, Performance and Upgrading.* Macmillan Publishing Co., Inc., New York.

Miescier, J. J. and Cabelli, V. J. (1982). Enterococci and other microbial indicators in municipal wastewater effluents. *Journal of the Water Pollution Control Federation* **54**: 1599-1606.

Moeller, J. R. and Calkins, J. (1980). Bactericidal agents in wastewater lagoons and lagoon design. *Journal of the Water Pollution Control Federation* **52**: 2442–2451.

MWD (1972). Lagoon treatment of farm wastes. Public health engineering section, Civil Division, Ministry of Works, NZ, .

Nakajima, J. and Kaneko, M. (1991). Practical performance of nitrogen removal in small-scale sewage treatment plants operated in intermittent aeration mode. *Water Science and Technology* **23**: 709-718.

Natarajan, K. V. (1970). Toxicity of ammonia to marine diatoms. *Journal of the Water Pollution Control Federation* **42**(5): R154-R190.

NTAC (1968). Water Quality Criteria: Report of the National Technical Advisory Committee to the Secretary of the Interior. Federal Water Pollution Control Administration, Washington DC.

NZAEI (1984). *Agricultural Waste Manual.* NZAEI, Lincoln College, Canterbury, New Zealand.

O'Brien (1975). *Algae removal by rock filtration*. Proceedings of the Water resources symposium No. 9. Ponds as a wastewater treatment alternative, University of Texas, Austin, University of Texas.

Oswald, W. J. (1968). Advances in anaerobic pond systems design. *Advances in Water Quality Improvement.* E. F. Gloyna and W. W. Eckenfelder Jr. Eds. University of Texas Press, Austin TX. 409-426.

Oswald, W. J. and Ramani, R. (1976). The fate of algae in receiving waters. *Ponds as a Wastewater Treatment Alternative* . E. F. Gloyna, J. F. Malina Jr. and E. M. Davis Eds. University of Texas, Austin TX. 111-122.

Pano, A. and Middlebrooks, E. J. (1982). Ammonia nitrogen removal in facultative wastewater stabilization ponds. *Journal of the Water Pollution Control Federation* **54**(4): 344-351.

Pearson, H. W., Mara, D. D. and Arridge, H. A. (1995). The influence of pond geometry on facultative and maturation waste stabilisation pond performance and efficiency. *Water Science and Technology* **31**(12): 129-139.

Pearson, H. W., Mara, D. D. and Bartone, C. R. (1987). Guidelines for the minimum evaluation of the performance of full-scale waste stabilization pond systems. *Water Research* **21**(9): 1067-1075.

Pedahzur, R., Nasser, A. M., Dor, I., Fattal, B. and Shuval, H. I. (1993). The effect of baffle installation on the performance of a single-cell stabilization pond. *Water Science and Technology* **27**(7-8): 45-52.

Polprasert, C. and Agarwalla, B. K. (1995). Significance of biofilm activity in facultative pond design and performance. *Water Science and Technology* **12**: 119-128.

Price, D. S., Smith, D. W. and Stanley, S. J. (1995). Performance of lagoons experiencing seasonal ice cover. *Water Environment Research* **67**(3): 318-326.

Qin, D., Bliss, P. J., Barnes, D. and FitzGerald, P. A. (1991). Bacterial (total coliform) die-off in maturation ponds. *Water Science and Technology* **23**(3): 1525-1534.

Quinn, J. M. and Gilliland, B. W. (1989). The Manawatu River clean-up: has it worked? *Transactions of the Institute of Professional Engineers of New Zealand* **16**(1): 22-26.

Quinn, J. M. and Hickey, C. W. (1993). Effects of sewage waste stabilization lagoon effluent on stream invertebrates. *Journal of Aquatic Ecosystem Health* **2**: 205-219.

Ray, D. E., Barkle, G. F., Selvarajah, N., McLeod, M. and Balks, M. R. (1995). *Seepage from dairy farm effluent treatment ponds.* Proceedings of the New Zealand Water and Wastes Association Annual Conference, Ellerslie Conference Centre, Auckland.

Reed, S. C. (1985). Nitrogen removal in wastewater stabilization ponds. *Journal of the Water Pollution Control Federation* **57**(1): 39-45.

Reed, S. C., Middlebrooks, E. J. and Crites, R. W., Eds. (1988). *Natural systems for waste management and treatment.* McGraw-Hill Book Company, New York.

Reynolds, J. H., Nielson, S. B. and Middlebrooks, E. J. (1975). Biomass distribution and kinetics of baffled lagoons. *Journal of the Environmental Engineering Division, American Society of Civil Engineers* **101**: 1005-1024.

Rich, L. G. (1980). *Low-maintenance mechanically simple wastewater treatment systems.* McGraw-Hill, New York.

Ritter, W. R. (1989). Odour control of livestock wastes: state-of-the-art in North America. *Journal of Engineering Research* **42**: 51-62.

RMA (1991). *Resource Management Act.* Government Printer, Wellington, NZ.

Rodenburg, J. (1985). Water quality can affect production and health. *Ontario Milk Producer* **62**(2): 12-14.

Romli, M., Keller, J., Lee, P. L. and Greenfield, P. F. (1994). The influence of pH on the performance of a two-stage anaerobic treatment system: model prediction and validation. *Water Science and Technology* **30**(8): 35-44.

Schierup, H., Brix, H. and Lorenzen, B. (1990). Wastewater treatment in constructed reed beds in Denmark— state of the art. *Constructed wetlands in water pollution control* . P. F. Cooper and B. C. Findlater Eds. Permagon Press, Oxford.

Schulz, T. J. and Barnes, D. (1990). The stratified facultative lagoon for the treatment and storage of high strength agricultural wastewater. *Water Science and Technology* **22**(9): 43-50.

Shin, H. K. and Polprasert, C. (1987). Attached-growth waste stabilization pond treatment evaluation. *Water Science and Technology* **19**(12): 229-235.

Shin, H. K. and Polprasert, C. (1988). Ammonia nitrogen removal in attached-growth ponds. *Journal of Environmental Engineering* **14**(4): 846-863.

Silva, S. A., de Oliveira, R., Soares, J., Mara, D. D. and Pearson, H. W. (1995). Nitrogen removal in pond systems with different configurations and geometries. *Water Science and Technology* **31**(12): 321-330.

Sinton, L. W., Donnison, A. M. and Hastie, C. M. (1993). Faecal streptococci as faecal pollution indicators: a review. Part I: Taxonomy and enumeration. *New Zealand Journal of Marine and Freshwater Research* **27**: 101-115.

Siuda, W., Wcislo, R. and Chróst, R. J. (1991). Composition and bacterial utilization of photosynthetically produced organic matter in an eutrophic lake. *Arch. Hydrobiologia* **1214**: 473-484.

Smith, R. E. (1983). Anaerobic lagoons- design and solids management. ASAE, 83-4068.

Stevenson, C. D. (1980). *Rationale for classification standards for BOD.* Proceedings of the Aquatic Oxygen Seminar Proceedings, Hamilton, National Water and Soil Conservation Organisation.

Stumm, W. and Morgan, J. J. (1981). *Aquatic chemistry.* 2nd ed., Wiley, New York.

Sukias, J. P. S. (1995). Aeration for preventing odours and improving quality of effluents from the DRC dairy sheds. NIWA, Consultancy Report DRC001.

Sukias, J. P. S., Davies-Colley, R. J., Tanner, C. C. and Hill, E. R. (1995). Mechanical aeration of dairy shed facultative waste stabilisation ponds. National Institute of Water and Atmospheric Research, Consultancy Report prepared for the Dairy Research Institute. DRI001. Hamilton, NZ.

Sukias, J. P. S. and Tanner, C. C. (1993). Investigation of constructed wetlands for tertiary treatment of piggery wastewaters. ARC Environmental Division, Technical Publication No.32. Auckland, N.Z.

Sukias, J. P. S. and Tanner, C. C. (1995). *Treatment of piggery waste stabilisation lagoon discharges in constructed wetlands*. Proceedings of the Wetlands for water quality control, Townsville, Qld, Australia.

Sukias, J. P. S. and Tanner, C. C. (1996). Farm-scale wetland treatment of dairy shed waste stabilisation pond discharges? NIWA, Consultancy report NZD001. Hamilton.

Sutherland, P. D. (1981). *Significance of sewage lagoon algae in receiving waters.* Proceedings of the Australian Water and Waste Association 9th Federal Conference, Perth WA.

Swanson, G. R. and Williamson, K. J. (1980). Upgrading lagoon effluents with rock filters. *Journal of Environmental Engineering Division, ASCE* **106**: 1111–1119.

Tanner, C. C., Clayton, J. S. and Upsdell, M. P. (1995). Effect of loading rate and planting on treatment of dairy farm wastewaters in constructed wetlands I. Removal of oxygen demand, suspended solids and faecal coliforms. *Water Research* **29**(1): 17-26.

Tanner, C. C., Clayton, J. S. and Upsdell, M. P. (1995). Effect of loading rate and planting on treatment of dairy farm wastewaters in constructed wetlands II. Removal of nitrogen and phosphorus. *Water Research* **29**(1): 27-34.

Tanner, C. C. and Sukias, J. P. (1995). Accumulation of organic solids in gravel-bed constructed wetlands. *Water Science and Technology* **32**(3): 229-240.

Uhlmann, D. (1980). Limnology and performance of waste treatment lagoons. *Hydrobiologia* **72**: 21-30.

USEPA (1976). Quality Criteria for Water. U S Environmental Protection Agency, EPA-440/9-76-023. Washington DC.

USEPA (1984). Process design manual for land treatment of municipal wastewater; supplement on rapid infiltration and overland flow. United States Environmental Protection Agency, EPA 625/1-81-013a. Cincinnati OH.

USEPA (1985). Ambient water quality criteria for ammonia 1984. Criteria and Standards Division, U S Environmental Protection Agency, . Washington D. C.

USEPA (1986). Quality Criteria for Water. U.S Environmental Protection Agency, EPA 440/5-86-001.

Valentis, G. and Lesavre, J. (1990). Wastewater treatment by attached-growth micro-organisms on a geotextile support. *Water Science and Technology* **22**(1/2): 43-51.

van Oostrom, A. J. and Russell, J. M. (1994). Denitrification in constructed wetlands receiving high concentrations of nitrate. *Water Science and Technology* **29**(4): 7-14.

Warburton, D. J. (1983). *Lagoon performance treating livestock waste*. Proceedings of the 15th New Zealand biotechnology conference, Massey University, Palmerston North, New Zealand.

Warburton, D. J. and Parkin, M. F. (1982). The performance of an anaerobic/aerobic lagoon after shock loading. *New Zealand Journal of Dairy Science and Technology* **17**: 177-184.

Warren, K. S. (1962). Ammonia toxicity and pH. *Nature* **195**: 47-49.

Welch, E. B., Quinn, J. M. and Hickey, C. W. (1992). Periphyton biomass related to point-source nutrient enrichment in seven New Zealand streams. *Water Research* **26**(5): 669-675.

WHO (1989). Health Guidelines for the Use of Wastewater in Agriculture and Aquaculture. World Health Organisation, Report Series 778. Geneva, Switzerland.

Wilcock, R. B., McBride, G. B., Nagels, J. W. and Northcott, G. L. (1995). Water quality in a polluted lowland stream with chronically depressed dissolved oxygen: causes and effects. *New Zealand Journal of Marine and Freshwater Research* **29**: 277-288.

Williams, A. G., Shaw, M., Selviah, C. M. and Cumby, R. J. (1989). The oxygen requirements for deodorizing and stabilizing pig slurry by aerobic treatment. *Journal of Agricultural Engineering Research* **43**: 291-311.

WPCF (1990). *Natural systems for wastewater treatment; Manual of Practice FD-16*. Water Pollution Control Federation, Alexandria VA.

Wrigley, T. J. and Toerien, D. F. (1990). Limnological aspects of small sewage ponds. *Water Research* **24**(1): 83-90.