Effects of biologically active discharges into aquatic ecosystems: Review of treatment systems and standards

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Abstract

Over the last ten years there has been an increasing awareness of the impacts of uncontrolled biological waste discharge into the receiving waters. The types of impact such discharges produce in the receiving waters are discussed. Early standards were based on overseas research but there is now a substantial body of New Zealand based work. Initial research concentrated on dairy shed discharges and domestic sewage pond discharges but has broadened to include piggeries wastes, meat industry wastes and various others.

In the main this work has shown that traditional oxidation pond systems do not treat the water to a high enough standard. Currently many discharges are high in suspended solids, biochemical oxygen demand, nitrogen (often in the form of ammonia), phosphorus and faecal coliforms.

The effectiveness of additional treatment in wetlands, either natural or constructed is being investigated. Early results are encouraging and suggest that low cost, low maintenance wetlands can be constructed to polish effluent from oxidation ponds. Research into other methods such as spraying onto land is continuing.

The dilution factors required to achieve various receiving water standards are presented for both domestic and dairy shed oxidation pond discharges.

Some wastes, such as those from piggeries, are proving difficult to treat to the required standard and work on this and other problems is continuing.

Department of Conservation staff involved in vetting resource consents for discharge to water involving biological wastes should consider recommending further polishing in wetlands.

1. Introduction

For many years water managers in New Zealand have relied upon overseas standards and data for setting local environmental standards. In some instances these have been totally inappropriate. For example, the recommended standard for suspended sediment to maintain salmonid fisheries values was 25 mg l^{-1} above ambient (Alabaster and Lloyd 1980). This may have been adequate for already polluted European waters, but work by Davies-Colley et al. (1992) has shown that values higher than 4 mg l^{-1} may bring about a 50% reduction in the stream invertebrates upon which fish depend for food.

USEPA guidelines (1985) suggest that ammonium toxicity to stream organisms may occur at levels as low as 0.52 gm^{-3}. Research by Hickey and Vickers (1994) shows that New Zealand stream invertebrates may be more sensitive by a factor of three than these figures indicate.

Conversely, there are some natural waters in New Zealand, most notably in the geothermal region, where parameters such as cadmium already exceed USEPA guidelines for protection of aquatic life.

Faced with such a variety of often conflicting information it is difficult for water managers and those commenting on proposed resource consent conditions to set scientifically justifiable standards. Fortunately research on the impacts of biologically active discharges has exploded in the last ten years and we are now in a position to suggest scientifically tenable conditions.

Areas of recent New Zealand research have included the performance of sewage oxidation ponds, the impact of sewage pond discharges on stream communities, the efficiency of various treatment regimes, the impact of sewage discharge on estuarine communities, efficiency of phosphorus removal, and sunlight inactivation of potential pathogens. These findings will be discussed under the appropriate headings and, where available, recommended maximum acceptable values in the receiving waters or the amount of dilution required are given in the appendices. The bibliography lists those papers which may be of interest but may not have been sighted by the author (cited but not sighted). Many of the papers quoted in this review have been written for a specialist audience. As a result acronyms, abbreviations and jargon abound. A glossary and a list of abbreviations (together with their amplification!) have been provided to help explain some of the more arcane terminology.

2. Aims of waste treatment

Biological wastes come from a variety of sources. Human sewage, dairy shed discharge, piggery wastes, meatwork effluent, fish processing plants, pulp mills, vegetable processing plants and even vineyard wastes can all bring about adverse environmental effects in receiving waters. Only after adequate treatment should process water or wastes be discharged.

In general, waste treatment aims to reduce the level of (James 1987):

i) biochemical oxidation demand (BOD)

ii) suspended solids

iii) nutrients

iv) pathogens

Ammonia can be added to this list as levels in some discharges may be toxic to stream organisms (Hickey *et* al. 1989a).

Obviously biochemical oxygen demand can be reduced by supplying oxygen while pathogens can be inactivated by sunlight (Davies-Colley *et al.* 1994). Removal of nutrients and suspended solids is more difficult. In many instances dilution may be the only available treatment.

Ideally, all discharges should be below levels likely to adversely affect receiving environment biota. However, this is unrealistic and the pragmatic approach (and that outlined by the Resource Management Act (RMA)) allows reasonable mixing and hence dilution.

3. Impact on receiving waters

Several recent New Zealand papers have focused on the impacts of human and dairy shed treatment pond discharges on the receiving waters. Hickey *et al.* (1989a) investigated the effluent characteristics of dairy shed oxidation ponds and evaluated their potential to impact on rivers. The effluents exhibited high biochemical oxygen demand, high suspended solids, and very high nutrient levels. Of particular concern were the ammonium levels (75 g m^{-3} NH_4-N) which represented a four-fold higher level of potential oxygen demand than the measured biochemical oxygen demand. Faecal coliforms were also high.

Quinn and Hickey (1992, 1993) investigated the ecological impacts of sewage oxidation pond effluent discharge to rivers. When dilutions were low (<15-fold) the downstream fauna was characterised by opportunistic species of small size and short generation time (mostly oligochaetes and chironomids). The Quantitative Macroinvertebrate Community Index (Stark 1985), a measure of the response of communities to elevated levels of pollutants, was lowered. In two streams, ammonia levels immediately below the discharge exceeded USEPA (1985) guidelines for protection of salmonids. Quinn and Hickey concluded that in some streams there were changes that represented "significant adverse effects on aquatic life".

In their 1993 paper Quinn and Hickey showed a marked reduction in insect predators where dilutions were <14-fold. However, total predator densities were not necessarily lowered. In two streams planarian densities were high below the discharge. The most sensitive indicator of the effect of effluent discharge was the percentage of common taxa which differed significantly between upstream and downstream sites. The authors also observed that where domestic sewage lagoons were mixed with industrial wastewaters, greater dilutions in the receiving water may be required.

3.1 BIOCHEMICAL OXYGEN DEMAND

Hickey *et al.* (1989a) suggest that oxygen depletion in the receiving waters is a possibility because of the high biochemical oxygen demand and the ammonia present in the discharge. However, most of the biochemical oxygen demand is likely to be due primarily to algae and particulate organic matter (Davies Colley *et al.* 1995). This results both from respiration and subsequent decay and may lower dissolved oxygen levels. Potential problems from this source have not yet been studied in detail in New Zealand.

Where ammonia levels are high the nitrifying bacterial population on the stream bed accounts for most of the nitrogenous oxygen demand (Hickey *et* al. 1989a). Accordingly rapid removal of NH_4-N may cause dissolved oxygen depletion.

3.2 AMMONIA

As well as creating the high oxygen demand referred to above, ammonia is extremely toxic to fish and aquatic invertebrates. Ammonia toxicity increases with increasing pH. As high ammonia levels are usually associated with high levels of other nutrients there are often algal blooms and/or proliferation of macrophytes in the receiving waters. Increased plant biomass leads to increased high pH excursions (because carbon dioxide is removed from the water which reduces the acidity). This increases the toxicity of the ammonia. Nitrification ameliorates high ammonia levels. Hickey *et* al. (1989a) suggest that multiple dairy shed discharges can lead to toxic ammonia levels. For a pond effluent diluted 100-fold, concentrations that would be potentially toxic to salmonids would be discharged around 50% of the time.

Davies-Colley *et al.* (1995) state that free ammonia (NH_3) may inhibit algal photosynthesis within lagoons at concentrations greater than about 10 g m^{-3}.

3.3 PATHOGENS

For many years the level of faecal coliform bacteria was measured to give an indication of disease risk associated with the water quality. The faecal coliforms themselves are not considered particularly dangerous but they were believed to give an indication of the number of viral and other pathogens likely to be found in the water. In other words, if faecal coliforms were high, so would the levels of more dangerous organisms be. It is now considered that levels of the enterococcus group appear to be a better predictor of the risk to swimmers of contracting gastro-intestinal illnesses (Cabelli *et al.* 1983 in Davies-Colley *et al.* 1994).

3.4 SUSPENDED SOLIDS

Suspended solids may also adversely impact on stream fauna. The ways in which these effects may occur are reviewed in some detail by Ryan (1991). Principal impacts come from lowered light penetration and the associated reduction in productivity. Other effects include the smothering of benthos by filling gaps in the substrate with settled material, lowering of the food value of periphyton, and reducing the hunting success of visual predators.

More recently, Davies-Colley *et* al. (1992) have quantified the level (4 mg l^{-1} above ambient) at which significant impacts occur in clear water streams. This figure was given further support by Quinn and Hickey (1993). They showed that more than half of the common taxa differed significantly in density downstream of discharges where the increase in suspended solids exceeded 4 g m^{-3}. Species normally considered sensitive to pollutants (mayflies, stoneflies and caddisflies) increased at low levels of increased suspended solids but declined to about 50% of the upstream values once the increase in suspended solids exceeded 4 g m^{-3}. This phenomenon (an increase in densities at low pollutant levels and a decrease at higher levels) is referred to as a subsidy-stress response (Odum 1985).

Quinn and Hickey also found that, at low to moderate dilutions, the zone of impact from the discharge extended from 1.3 to 4.5 km downstream.

3.5 NUTRIENTS

Elevated nutrient levels (particularly phosphorus and nitrogen) can stimulate both periphyton and rooted macrophyte growth. Lake Rotorua, for many years the recipient of sewage discharge, provides a classic example of the effects of nutrient enrichment. The weed beds which have resulted are still a source of concern to locals because of the undesirable aesthetic effects (smell from rotting vegetation, windrows on beaches, difficulty in swimming, etc). Cooper (1994) discusses a combined land/wetland treatment regime for the Rotorua sewage.

Algal slimes are also of concern. They may not be particularly palatable to grazer species and can grow relatively unchecked, they slough off at regular intervals and clog water intakes, and they may also cause big fluctuations in both dissolved oxygen and pH (Quinn and Gilliland 1989). In January 1979, severe nighttime depletion of dissolved oxygen, attributable to algal and sewage fungus respiration, killed hundreds of trout in the Manawatu River. A subsequent episode of lowered nighttime oxygen levels in 1984 even killed eels, normally amongst the hardiest of fish. The authors recorded pH levels as high as 9.8 after changes to the benthic community structure. High pH increases the toxicity of ammonia.

Algal slimes are also aesthetically unacceptable; few people like to swim or fish where there is a filamentous algal bloom.

In estuaries or enclosed harbours nutrient enrichment can bring about algal blooms. Vant and Larcombe (1994) detail the apparent effects of nitrogen enrichment on planktonic biomass (measured as chlorophyll *a* in mg m^{-3}). They note seasonally elevated levels of phytoplankton in Manukau Harbour and correlate this with nitrogen in the Manukau Wastewater Treatment Plant discharge. Nutrient levels and phytoplankton concentrations were considerably lower elsewhere in the harbour.

Welch *et al.* (1992) specifically investigated the impact of nutrient enrichment on stream periphyton. They found increased periphyton biomass in response to point source enrichment in 4 of the 7 streams and rivers studied. However, the model they developed predicted greater algal growth than was actually observed in many instances. The authors conclude that shading by riparian vegetation, unsuitable substrate and high invertebrate grazer densities prevented growth achieving the levels the nutrient enrichment would indicate.

Sewage fungus may occasionally occur, usually where there are high levels of filterable biochemical oxygen demand (fBOD-biochemical oxygen demand that passes through a filter in a given period of time). This can promote sewage fungus, particularly where $fBOD_5$ levels exceed 2 g m^{-3} (Ministry for the Environment 1992). Quinn and GilIiland (1989) found that increases of dairy waste induced $fBOD_5$ below 1.8 g m^{-3} limited sewage fungus growth in the Manawatu River. Davies-Colley *et al.* (1995) showed that the median $fBOD_5$ for 11 domestic sewage oxidation ponds was only slightly greater than the guidelines for receiving water quality.

4. Domestic sewage ponds

Hickey *et al.* (1989b) investigated the characteristics of 18 domestic sewage ponds. These were all designed to national specifications; namely a design load of 84 kg biochemical oxygen demand (BOD) ha^{-1} day^{-1} for the primary pond. A wide range of parameters was measured monthly for at least a year in three regions (Auckland, Manawatu, and Southland).

Despite most of the ponds having loadings within their design criteria, they varied widely in their efficiency. No significant relationships were found between effluent quality and the percentage of design influent load. Furthermore few relationships with retention time were identified.

Biological oxygen demand, suspended sediment and faecal coliforms did not appear to fluctuate seasonally. Ammonium (NH_4-N) showed increased concentrations in winter, probably as result of reduced activity by temperature sensitive nitrifying bacteria.

The authors point out that the sampling programmes from which the data were obtained were not designed to identify processes occurring within the ponds. Light and wind, which would have had a major impact on pond dynamics, were not measured. Furthermore, the monthly sampling interval would be longer than many of the component processes. For instance, dissolved oxygen varied diurnally by as much as 20 g m^{-3} in response to changing light; rapid algal growth may result in marked clarity reductions (and presumably an increase in suspended material) in periods of less than a week. Wind can bring about marked clarity changes in a matter of minutes, as a result of resuspension.

While processes may not have been identified, the quality of the effluent and the degree of dilution required to protect instream values were. Table 4 in their paper summarises the effluent concentration and the dilution factor required and has been reproduced here, suitably modified as Appendix 1.

Hickey and Quinn (1990) in further analyses of the data gathered in the above study confirmed that there was no relationship between pond loadings and biochemical oxygen demand, suspended solids, ammonium, dissolved oxygen or faecal coliforms. Similarly, despite the 11-fold variation in retention time, the only relationship revealed was a tendency for declining concentrations of ammonium and increased levels of dissolved oxygen with increased retention time.

Hickey and Quinn also compared the performance of single pond and two pond systems in reducing median effluent concentrations. Because residence times varied widely it was only possible to make comparisons between a few ponds. In systems with 25-50 day retention times single-stage ponds exhibited higher ammonium and lower dissolved oxygen levels than two-stage ponds. There appeared to be no difference in suspended solids levels.

Assessment of operational efficiency presents problems. Reduction of the oxygen demand of the influent material occurs, but with the production of new organic matter in the form of algal cells. The cells represent a substantial biochemical oxygen demand and growth within the ponds may increase the BOD. The authors point out that this phenomenon makes it difficult to assess the relationship between load and BOD. Algal growth may also increase the amount of suspended solids, which cuts down light penetration and the effectiveness of the pond in inactivating potential pathogens. This phenomenon is explored at some length by Davies-Colley et al. (1995).

Hickey and Quinn (1990) concluded that, apart from lowered ammonia concentrations, there were no significant improvements (in terms of biochemical oxygen demand, suspended solids and faecal coliform concentrations) by increasing retention time. They suggest that design improvements are necessary to improve the operational efficiency of ponds.

Davies-Colley *et al.* (1995) studied the effluent characteristics of 11 domestic sewage waste stabilisation ponds designed to national specifications. The investigation emphasised optical properties, biochemically active constituents and receiving water impacts. Twenty-six variables were measured and compared, where applicable, with findings from earlier studies.

As with previous studies, Davies-Colley *et al.* found a considerable range in effluent composition with high nutrient levels at all times of the year. The ratio of filterable BOD_5 to total BOD_5 appeared to be a useful indicator of lagoon efficiency with high levels indicating poor performance (see next paragraph for an explanation of this observation). Algal photosynthesis was frequently limited by restricted light penetration due to high algal biomass and aquatic humus (yellow substance). Pond effluent character appeared to be strongly influenced by sunlight and wind.

The authors did find a positive correlation between residence time and suspended solids. Somewhat tongue in cheek they comment that longer retention times function mainly to grow greater biomasses of algae! Elevated ammonia levels were measured in systems that were overloaded or industrially-loaded. In such lagoons a negative feedback process may operate; algal photosynthesis raises pH and free ammonia rises to levels toxic to the algae. Lowered production of photosynthetic oxygen results in less rapid microbial oxidation of soluble organics with the result that filterable BOD_5 in the effluent is a greater proportion of total BOD_5.

Lagoon effluents were high in yellow substance, at least in comparison with 96 New Zealand rivers. According to Curtis *et al.* 1994 (in Davies-Colley et al. 1995) yellow substance in illuminated lagoon water photosensitises the sunlight inactivation of enteric bacteria. This apparently occurs through the action of photochemical transients such as singlet oxygen, superoxide, or hydroxyl radicals. High daytime pH and dissolved oxygen levels produced by high algal biomasses tend to compensate for the reduction in light penetration caused by the algae (with respect to inactivating enteric bacteria).

Davies-Colley *et al.* (1995) conclude that sewage lagoons need to be upgraded to reduce bacterial, NH_4-N and suspended solids levels.The variability in performance also needs to be addressed. Additional shallow polishing ponds may need to be added as well as baffles within lagoons to improve flow characteristics and prevent short-circuiting. There may also be a need for further treatment by natural or artificial wetlands or land application to avoid significant adverse impacts on receiving waters.

5. Dairy shed treatment ponds

To date the only comprehensive study of the effectiveness of dairy shed oxidation pond systems has been reported in the paper by Hickey *et al.* (1989a). Eleven dairy shed oxidation ponds, nominally built to national specifications, were examined for a range of parameters. Wide variations were noted between ponds. Suspended solids varied 9-fold while biochemical oxygen demand varied 3-fold. Most ponds exhibited high levels of ammonium, dissolved reactive phosphorus and, not surprisingly, faecal coliform bacteria.

Although loadings in these two pond systems were, with one exception, below recommended for the first (anaerobic) pond and below or 5% over for the second (aerobic) pond, they were relatively inefficient at removing suspended solids, reducing biochemical oxygen demand, or ammonia levels. Hickey *et al.* point out that dairy pond effluent compares badly with effluent from domestic sewage oxidation ponds. Dairy ponds had higher median biochemical oxygen demand (by 4-fold), suspended solids (by 5-fold) and ammonia (by 11-fold) than domestic sewage ponds. It is apparent from this that the performance of dairy shed oxidation ponds needs to be improved, perhaps by further polishing in natural or constructed wetlands. Some of the recent papers on wetland treatments (e.g. Tanner 1994) are discussed at length in the next section.

6. Wetland treatment of oxidation pond effluent

6.1 NATURAL WETLANDS

Oxidation pond effluent may be treated by passage through natural or artificial wetlands. The performance of wetlands at removing substances usually diminishes with time although this is not always the case. Cooke (1992, 1993, 1994) and Cooke *et al.* (1990, 1992) investigated the effects of sewage effluent from the resort town of Paihia on a natural wetland in the Waitangi forest (Northland). Comparisons were able to be made with an adjacent wetland which acted as a control.

The results suggested that the treatment wetland was capable of removing very large quantities of nitrogen and phosphorus. Soil in the sewage wetland contained ammonium and phosphorus two orders of magnitude greater than that of the reference wetland (Cooke *et al*., 1990). The high phosphorus deposition stemmed from reactions between the discharge and the low pH, high iron and aluminium wetland waters. Most dissolved reactive phosphorus was removed from the system and deposited in the sediments but was often remobilised by high flow rates. Overall there was between 34 and 85% removal of dissolved reactive phosphorus (DRP) and 28 and 70% reduction in total phosphorus (TP) during the four 4-day sampling events. The study also concluded that P removal by wetland vegetation only accounted for an 8% reduction and argued that most of this phosphorus would have been derived from the underlying sediments. Annual die-back of the dominant bulrush *Typha orientalis* would release most of this P back into the system (Cooke 1992).

Cooke (1994) concludes from this, and his previous studies, that if P removal is a major objective of treatment, then a high loading into a small natural wetland (which could be constructed) also receiving drainage from mineralised wetlands, may be more cost-effective and sustainable than lightly-loading a large but hydrologically-isolated wetland system.

Nitrogen inflows were either in predominantly ammonium or nitrate form (Cooke 1994) depending upon prevailing meteorological conditions. In summer most of the nitrogen was in nitrate form and it was all transformed during passage through the wetland. Cooke's isotope studies indicated that ~ 60-70% was denitrified, 25-35% converted to ammonium and 5-10% assimilated. Despite this, considerable amounts of nitrate were exported from the wetland especially during spring-early summer high flows. Assays on sediments showed a marked increase in nitrification activity at the confluence with the natural wetlands. Changes in redox potential at these sites provides ideal conditions for nitrification of sorbed ammonium which is subsequently flushed from the system in flood events.

Removal of nitrogen during flood events will dilute the concentration and speed its removal from the stream system. Despite the nitrification, the wetland is still able to ameliorate the effects of increased nutrient loading.

6.2 CONSTRUCTED WETLANDS

Because of the size of New Zealand's domestic animal herd and associated wastes from processing plants, considerable effort has been put into effluent treatment research. Bhamidimarri *et al.* (1991) summarise the New Zealand experience to that date. Sukias and Tanner (1994) discuss the types and mode of action of constructed wetlands. In both surface flow and sub-surface flow wetlands the substrate and lower stems and roots of plants act as attachment points for bacteria and other micro-organisms. Suspended solids and dissolved organic matter are retained in the wetland by settling and filtration. Micro-organisms break down much of the trapped material while organic nitrogen and ammonia are converted to less environmentally active forms such as nitrogen gas or nitrate.

Wetlands may be used in conjunction with existing pond systems, have relatively low maintenance and can treat waste water to a high level. The cost varies depending upon the availability and cost of construction materials.

An excellent summary article on constructed wastewater wetlands has been written by the Meat Industry Research Institute of New Zealand (MIRINZ 1994). The institute used surface flow wetlands to treat meat processing effluent with reasonable success. When the inflow was from secondary aerobic ponds and a supply of carbon was available, nitrogen removal was as high as 20-40%. When inflow was from anaerobic ponds, about half this level was removed. The institute concluded that surface-flow wetlands were more economical to operate than sub-surface flows and achieved nearly the same performance level.

Most of the information for the MIRINZ bulletin came from work by van Oostrom and Russell (1994). Laboratory experiments utilising giant sweet grass *Glyceria maxima* floating on a nitrified meat processing effluent showed an ability for the system to denitrify 3.8 g m^{-2} of NO_3-N per day at 20° C. In a pilot-scale wetland covered with sweet grass and receiving a nitrified meat processing effluent, nitrogen removal rates ranged from 0.6 g m^{-2} per day in winter to 3.0 9 m^{-2} per day in summer. The floating plant mat and the sediment were the most active denitrification sites.

Even higher nitrogen removal rates were measured by van Oostrom (in press) using four small-scale surface-flow wetland treatments. Three of these utilised sweet grass while the fourth used a simulated plant mat constructed of nylon fabric. The planted wetlands removed about twice that of the unplanted wetland and averaged 46-49% nitrogen removal. Summertime nitrogen removal reached 75% (approximately 9.5 g m^{-2} d^{-1}). About 87% of the nitrogen removed was due to denitrification, with the remaining 13% due to accumulation in sediment and plant biomass. The plants were responsible for about 50% of nitrogen removal, mainly through the supply of organic carbon and creating anaerobic conditions for denitrification.

Tanner (1994) investigated the relative efficiencies of horizontal and up-flow gravel bed constructed wetlands in treating dairy farm wastewaters. These were both sub-surface flow treatments in which the effluent flowed through gravel, not over the top. With longer retention, more BOD, TN and TP were removed. There was little difference between formats in their ability to reduce BOD (70-90% reduction) and SS (40-90%) reduction. However the horizontal flow wetlands which reduced TN by 40-90% and TP by 30-80% were more efficient at this than the up-flow wetlands. Phosphorus removal was minimal except in the longest retention time trials and declined during the trial. Tanner concludes that plant uptake and storage, and sediment adsorption capacities of the wetlands were exceeded at the loading rates tested and that the levels of P removal were unsustainable in the long term.

Tanner *et al.* compared the effectiveness of planted and unplanted horizontal flow wetlands in removing oxygen demand, suspended solids, faecal coliforms (1995a), nitrogen and phosphorus (1995b). At the highest loading tested, carbonaceous biochemical oxygen demand ($CBOD_5$) removal was significantly lower in the unplanted than in the planted system. Planted wetlands generally showed greater BOD removal than the unplanted ones, particularly at higher loadings. Both achieved similar efficiencies in removal of suspended solids: over 75% at all loading rates. Faecal coliforms were removed at rates from 90-99% (increasing with increased retention time). Tanner *et al.* conclude that, at the mean levels of faecal coliforms recorded in the effluent, livestock drinking water guidelines could be met in receiving waters at moderate levels of dilution (>15 to 40 fold).

Both planted and unplanted wetlands showed increased TN and TP removal with increased retention times. However, the unplanted wetlands showed a marked decline in TN and TP removal at high loadings. Storage by plants accounted for between 3 and 20% of the N removal and between 3 and 60% of the P removal in the planted wetland. The authors suggest that the balance presumably came from other plant-associated processes.

They conclude that constructed wetlands during their first two years of operation can remove considerable quantities of N and P from ammonia-rich dairy farm wastewaters. Treatment levels for the planted wetland suggest that direct ammonia toxicity and nitrogenous demand for the discharges would have been reduced 1.5-3.5 fold with increasing retention time and total phosphorus reduced 1.6-3.8 fold.

Individually, piggery wastes pose a considerably greater threat to receiving waters than dairy shed effluent. Sukias (1993) quotes levels of BOD between 100 - 600 g m^{-3} (Vanderholm 1985) and Oleszkiewicz and Koziarksi (1981, in Sukias 1993) state that the wastes contain high levels of ammoniacal nitrogen. In 1989 a joint research project between NIWA and the Auckland Regional Water Board was initiated. The study investigated the use of constructed wetlands to treat piggery shed effluent. Despite problems with continuity (operating regimes and management changed and then one piggery closed) some preliminary information was obtained. Wetlands removed 50% of SS in the South Auckland Piggeries study but only 37% in the Paerata study. Both sites removed around 60% of the BOD but that remaining appears to be too high for discharge into streams. P was removed at a level of around 20% while TN removal was 37% at South Auckland and only 12% at Paerata. High pH values at South Auckland may have led to volatilisation of ammonia. Contrary to results from overseas studies which indicate removal rates of 90 - 99%, Sukias found little removal of faecal coliforms. Sukias and Tanner (1994) note that some piggery effluent may contain ammonia levels in excess of 800 g m^{-3}. To protect instream values dilution factors of over 4000 would be required.

Sukias and Tanner (1995) investigated three methods of treating piggery wastes: laboratory scale vertical flow cascade systems, horizontal flow constructed wetlands and a full-scale constructed wetland with both surface and subsurface flow sections. The vertical flow cascade system consisted of a series of five buckets filled with gravel and arranged so that the overflow from the top one entered the bottom of the next. Effluent was introduced twice daily to the bottom of the top bucket.

Treatment efficiency varied depending upon factors such as retention time, pig food and piggery management procedures. Rates of BOD_5 and SS removal were related and low. It appears that pig effluent has low settleability in comparison with dairy shed wastes; this, in conjunction with inadequate pre-treatment reduced the performance of the systems for BOD_5 and SS removal. However, rates of SS removal of around 40-50% were achieved with BOD_5 varying from 42% in the full scale scheme to 70% in both the cascade system and the horizontal flow wetland systems.

It is evident that further trials on piggery wastes are necessary, probably involving more adequate pre-treatment before polishing in the constructed wetlands. As piggery wastes can require the extraordinary BOD_5 of up to 30,000 g m^{-3} (NZAEI 1984 in Sukias and Tanner 1995) the problem does not lend itself to an easy solution. Further research on the subject is continuing.

With the obvious exception of piggery wastes (for the reasons outlined above), in situations where natural wetlands are not available, or it is considered inappropriate to use them, constructed wetlands have the potential to polish effluent quality to a point where only small dilutions in the receiving water are required.

7. Conclusions

Piggery wastes pose a major threat to aquatic systems. Generally neither dairy shed oxidation ponds nor domestic sewage ponds treat their discharges to an adequate standard. Further treatment to reduce levels of total phosphorus, total nitrogen, biochemical oxygen demand, suspended solids and faecal coliforms is often required. Recent research into polishing effluents from such sources in natural or artifical wetlands show great promise and should be encouraged where possible by DOC staff who deal with resource consent applications. Where there are low pH wetland outflows rich in iron and aluminium, precipitation of phosphorus (and presumably heavy metals) in sewage and other discharges will occur on mixing.

At present there is little available information on impacts in receiving water from other sources such as fish factories or vineyard wastes. These discharges would probably also benefit from further polishing.

It is also evident that vigilance is required even where oxidation ponds are installed. As discussed above, they do not always treat water to an adequate standard. In such situations alternative disposal methods should be investigated, including, if suitable areas are available, spraying onto pasture or plantation forests. Cooper (1994) suggests that the two methods should complement each other well. Nitrification in the forest should lead to groundwater arriving at the wetland in a diffuse band. This nitrate-rich water will then react with wetland sediments, where denitrification should occur. Cooper warns that flows within wetlands are such that only a small proportion of reaction sites is actually reached by water flow. In other words the area actually available for removal processes may be quite small. More diffuse flow through a wetland may need to be encouraged.

Dilution levels for a variety of pollutants are listed in Appendices 1 and 2. These deal with dairy shed waste and domestic sewage pond discharge respectively and are only slightly changed from Hickey et al. (1989a, 1989b). There is insufficient information on treatment levels for meatworks and piggery waste to provide further guidance.

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Glossary

Aerobic - *utilising oxygen*
Anaerobic - *without oxygen*
Benthos - *the organisms on the bottom of a water body*
Biochemical oxygen demand (BOD). *the oxygen required by organisms for respiration and for the oxidation of chemicals*
Macrophytes - *rooted aquatic plants*
Nitrifying bacteria - *bacteria which convert free ammonia to nitrate and nitrite to nitrate*
Periphyton - *the organisms which grow over stones in a stream. May include algae, bacteria, fungus, etc.*
Taxa - *any forms of life (singular - taxon)*

Abbreviations

BOD - *biochemical oxygen demand*
CFU - *colony forming units*
DIN - *dissolved inorganic nitrogen*
DO - *dissolved oxygen*
DRP - *dissolved reactive phosphorus*
FC - *faecal coliforms*
SS - *suspended solids*
TN - *total nitrogen*
TP - *total phosphorus*

Appendix 1

Approximate dilutions required to protect instream values from domestic sewage oxidation pond effluent. (Adapted from Hickey et al. 1989b.) DO, dissolved oxygen; BOD biochemical oxygen demand; SS, suspended solids; DRP, dissolved reactive phosphorus; NH_4-N, ammonium nitrogen; NO_3-N, nitrate nitrogen; DIN, dissolved inorganic nitrogen.

Variable	Effluent concentration ($g\ m^{-3}$)		Receiving water criterion ($g\ m^{-3}$)	Dilution factor required		Use
	Median	95 Percentile		Median	95 Percentile	
DO	8	0.9	5 (min)	-	>6	biota respiration
BOD	27	70	5	>6	>11	biota respiration
SS	56	171	4 above background*	>14	>42	protect biota aesthetics
DRP	5.0	9.5	0.010 (max)	>500	>950	algal growths
NH_4-N	7.0	29	0.18**	>38	>161	stream invert. toxicity
NH_4-N (oxygen demand)	46	196	5 (DO)	>9	>39	biota respiration
NO_3-N	0.11	2.9	0.080 (max)	>1.5	>36	algal growths
DIN	7.1	32	0.080	>90	>400	algal growths
BOD and NH_4-H (oxygen demand)	46	196	5 (DO)	>9	>39	biota respiration
Faecal coliforms	4300	230 000	200	>22	>1100	bathing
			2000	>2	>115	drinking

* Davies-Colley et al. 1992, ** Hickey and Vickers 1994.

Appendix 2.

Approximate dilutions required to protect instream values from dairy shed oxidation pond effluent. (Adapted from Hickey et al. 1989a.) DO, dissolved oxygen; BOD biochemical oxygen demand; SS, suspended solids; DRP, dissolved reactive phosphorus; NH_4-N, ammonium nitrogen, NO_3-N, nitrate nitrogen; DIN, dissolved inorganic nitrogen.

Variable	Effluent concentration ($g\ m^{-3}$)		Receiving water criterion ($g\ m^{-3}$)	Dilution factor required		Use
	Median	95 Percentile		Median	95 Percentile	
DO	2.8	1.1	5 (min)	>1.8	>4.5	biota respiration
BOD	98	241	5	>20	>50	biota respiration
SS	198	804	4 above background*	>49	>201	protect biota aesthetics
DRP	12.2	17.1	0.010 (max)	>1220	>1710	algal growths
NH_4-N	75.0	191	0.18**	>416	>1061	stream invert. toxicity
NH_4-N (oxygen demand)	324	827	5 (DO)	>65	>165	biota respiration
NO_3-N	0.065	25.4	0.080 (max)	>1	>318	algal growths
DIN	75	216	0.080	>940	>2700	algal growths
BOD and NH_4-H (oxygen demand)	413	1068	5 (DO)	>83	>214	biota respiration
Faecal coliforms	70 000	540 000	200	>350	>2700	bathing
			2000	>35	>270	drinking

*Davis-Colley et al. 1992, **Hickey and Vickers 1994.