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PERTH, WESTERN AUSTRALIA

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All statements of fact and opinion in these proceedings are those of the authors. Authors are responsible for due acknowledgements and references in their papers. Neither the Conference Organisers or the editors shall be responsible for opinions expressed in the papers. All the papers were refereed and revised as necessary before publication.

K. Mathew & G. Ho
Editors

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FOREWORD

The International Environmental Technology Centre (IETC) is an integral part of the Division of Technology, Industry and Economics (DTIE) of the United Nations Environment Programme (UNEP). Its role is to promote the adoption and use of environmentally sound technologies (ESTs) for the management of urban and freshwater basin environmental issues in developing countries and countries in economic transition. ESTs are recognised as critical to success in achieving sustainable development.

One of the Centre's activities is to enhance the capacity of the decision and policy makers to make sensible technology choice. Adopting, applying and operating ESTs are key elements in policy-making and technology practice for integrated environmental management.

A global workshop on adopting, applying and promoting of environmentally sound technologies was organised by the UNEP-IETC in Dresden, Germany in 1996. The program combined proactive sessions, exercises, small group works, and field trips to build individual and institutional capacities: it focussed on priority issues such as freshwater resources, water supply, technology transfer, waste management and city planning.

In 1997 December UNEP-IETC sponsored a conference in Environmental Technologies for Wastewater Management at Murdoch University in Australia. Thirty participants from 19 countries were sponsored to participate at the conference.

The conference was targeted at the needs of national/local government, and civil-society decision-makers in the area of technology implementation/regulation, as well as at those with responsibilities in environmental management.

This publication contains papers selected from the papers presented at the conference after a thorough review process. As such it is a valuable source document for all who are involved in working towards sustainable development.

Lilia GC. Casanova
Officer-in-Charge/Deputy Director
UNEP-IETC

December 1999

EDITORIAL

The Environmental Technology Centre of the Institute for Environmental Science at Murdoch University was honoured to host the UNEP International Regional Conference on Environmental Technologies for Wastewater Management. There was an excellent response to the invitation to present papers with 33 papers from 11 countries included in the program. These papers were the basis of discussion and an interchange of ideas between practitioners, regulators, researchers and educators at the conference. After the conference they were peer-reviewed and revised, so that they will also become a resource for those wishing to find out about the state of the art in wastewater management technologies which are environmentally sound.

The papers and ideas presented therein were particularly useful to participants/decision makers from the Asia Pacific region attending the UNEP International Environmental Technology Centre (IETC) sponsored Workshop on Adopting, Applying and Operating Environmentally Sound Technologies for Urban Management held in conjunction with the conference and extending to 13 December 1997. Providing environmentally sound technologies in many countries in the Asia Pacific region is a huge challenge because of the high population density, difficulty in installing a piped sewerage system and other infrastructure and resource limitations.

Making an event such as the above happen required the support and help of many dedicated people. I would like to particularly thank Dr Christian Strohmann of the UNEP IETC who was consistent in his support and prompt in keeping with communication at all stages of the organisation of the conference and the workshop. Within the Institute for Environmental Science I was greatly assisted by personnel from the Remote Area Developments Group and the Environmental Technology Centre. I would like to especially mention Dr Kuruvilla Mathew, who was tireless in ensuring that details were attended to, and there were a myriad of them in the management of a conference such as this one.

I would like to express my appreciation to all authors of papers for contributing their ideas and the results of their research and experience, and also to those who contributed through discussion and through questions and challenging of ideas. Out of such interaction will theory and practice be advanced.

The conference was beneficial to all who participated, not only because of the meeting of ideas, but also because of the networking and friendships that were developed.

All the papers presented were subsequently reviewed and the authors were given an opportunity to make any necessary corrections. The selected papers are published in this proceedings. I would like to thank Mr Arpad Kolotas and Mr Nathan Malin for conducting the review process. I also would like to thank the authors and the panel of experts who conducted the review. I hope the publication will add to the knowledge in the practice of environmental technologies.

Goen Ho
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December 1999

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KEYNOTE SPEECHES

A VISION FOR THE FUTURE OF WATER REUSE: UNEP International Regional Conference on Environmental Technologies for Wastewater Management

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ABSTRACT

Reuse of wastewater is attracting wide community advocacy but support is languishing with any of the water utilities. The need to pioneer and finance reuse initiatives are proving too daunting for many suppliers, despite the crippling costs of conventional sewerage infrastructure investment and replacement. Here we explore some opportunities for reuse that address community aspirations for more efficient use of water resources. In particular, we perceive that the opportunity to promote reuse lies in consulting and educating our customers. Reuse also presents unprecedented opportunities for the environmental industry. Both ACTEW Corporation and the Environmental Management Industry Association are at the forefront of the recycling revolution.

ACTEW Corporation has consulted with its customers over a wide range of water initiatives including recycling; our experience is that major new initiatives require community support and genuine partnering.

KEY WAORDS

Wastewater, reuse, water reuse, community consultation, community involvement, EMIAA, ACTEW Corporation

INTRODUCTION

There has been a renaissance in community attitudes towards reuse of renovated wastewater. Almost everyone seems to believe that reusing wastewater is a good idea. When the NHMRC *Guidelines for use of reclaimed water in Australia* were released in 1987, the major emphasis was one of permitting reuse under a regime of public health assurance. The focus was on opportunistic demand - typically for watering golf courses with ponded effluent. The scope for reuse was visionary, but issues such as *resource use optimisation* were not even dreamt of.

New National Water Quality Management *Strategy Draft guidelines for sewerage systems -Use of reclaimed water* have attracted wide attention, particularly in their placing into context other facets of sewerage systems: acceptance of trade waste, effluent management, sludge management and sewerage system overflows. The new guidelines have a broad focus and stress reuse as an integral component of the total water cycle. The guidelines also cover public consultation, legal issues, forms of agreement for reuse, treatment, safeguards and control, and monitoring and reporting.

This forum examines reuse in the context of the urban water cycle. The urban water cycle has been a topic of foresighting analysis promoted by the Department of Industry, Science and Tourism (DIST) and has also been a topic of much wider debate surrounding integrated catchment management (ICM) and total water cycle management.

Before turning to the matter of placing wastewater in a broad environmental mental context, let us look briefly at some of the practical operating boundaries to existing pilot reuse projects. These boundaries can be characterised in terms of *scale of reuse, scope of technology, scope of application,* and scope for *community involvement.*

SCALE OF REUSE

The *scale of reuse* projects is not tied to specific water treatment technology. That said there are special demands of compactness and reliability required for applications such as major building and hotel recycling systems. The scale of reuse projects envisioned at this stage range from:

- small scale, eg individual residences or small groups of dwellings
- medium scale, eg the ACT Southwell park municipal landscape watering experiment
- large scale, eg Canberra's auxiliary sewerage treatment plant at Fyshwick with plans to water the Parliamentary triangle, irrigate pasture and turf on Dairy Flat, and augment the Duntroon playing fields watering system.

An important point to note about scale is that large numbers of domestic scale units may have a far greater impact on water use optimisation than single large projects. Indeed the potential for reuse appears to be greatest at the small scale end of the spectrum where the user is immediately in control.

SCOPE OF TECHNOLOGY

One of the important strategies ACTEW Corporation is focusing on is a diversity of technologies for renovated water reuse. The *scope of technologies* being investigated by ACTEW Corporation is nevertheless quite wide. Some of the more high technology solutions to water treatment may well have marketable applications well beyond Canberra, while the maturing market for small scale systems demand a very moderate scale technology, one that will cost less than a conventional sewer. Typically the domestic units with storage and in-built resource management system cost around 12,000 dollars. A conventional connection to sewer costs at least 8-15 thousand_dollars more per residence.

To a large extent the choice of water treatment technology depends on the reuse application and on physical location. Reuse schemes that can tap into existing conventional wastewater treatment plants are limited by pumping and wastewater transfer infrastructure costs. Pumping waste water more than a few kilometres is nearly always too costly. WATERMINING®, extracting water from a sewer, on the other hand lends itself well to built-up urban and open space watering opportunities. Broadly the levels of technology can be summarised as:

- simple aeration domestic scale systems;
- optimised conventional systems such as exist at Queanbeyan and Fyshwick; and
- high-technology packages such as the Better Cities Southwell Park and Cranos™ systems.

SCOPE OF APPLICATION

Simple availability of wastewater limits *the scope of applications.* It is doubtful, for example, that the water mining process (WATERMINING®) that exists at Southwell Park could ever meet all the demands of open space watering for a city like Canberra. At that location, seasonal demands of the playing fields, the nearby golf and race courses, and the showgrounds far exceed the existing sewer supply. Some innovations, such as storage in ornamental lakes and augmentation with stormwater, may be one solution, but such options are yet to be canvassed fully. Any consideration of reuse in isolation from total urban water cycle demands will result in poor realisation of the potential for optimising water use. Indeed there is likely to be some *fuzzing up* of the boundaries between different types of second class water in the scramble to find the most economic solution to applications such as irrigation. The problems for fully harnessing the so-called waste stream (it is all water) are multiplied many times in the larger cities.

A simple categorisation of level of use can nevertheless be made by analysis of specific site demands:

- low demand - domestic scale/ riverside irrigation of picnic grounds;
- medium scale demand - playing fields and open space watering; and
- high level demand - STP's servicing horticultural and major public area irrigation.

SCOPE FOR COMMUNITY INVOLVEMENT

ACTEW conducted a Community Reuse Roundtable over two days in June 1996. its purpose was to provide participants an opportunity to define their interest in, and commitment to, involvement in reuse projects. ACTEW, for its part, indicated a keenness to partner reuse projects with other agencies and the community as an intelligent outworking of principles espoused in the National Water Quality Management Strategy.

Likely participants for future reuse projects must include:

- operational groups;
- public agencies and groups with environmental leanings; and
- municipal agencies with public interest input such as health departments and EPA's.

Community involvement in reuse projects is also dictated by a number of physical factors including:

- cost of water (renovated wastewater versus costs of potable and stream or storm water);
- community education and consultation (see diagram); and
- agency interest in partnering reuse projects.

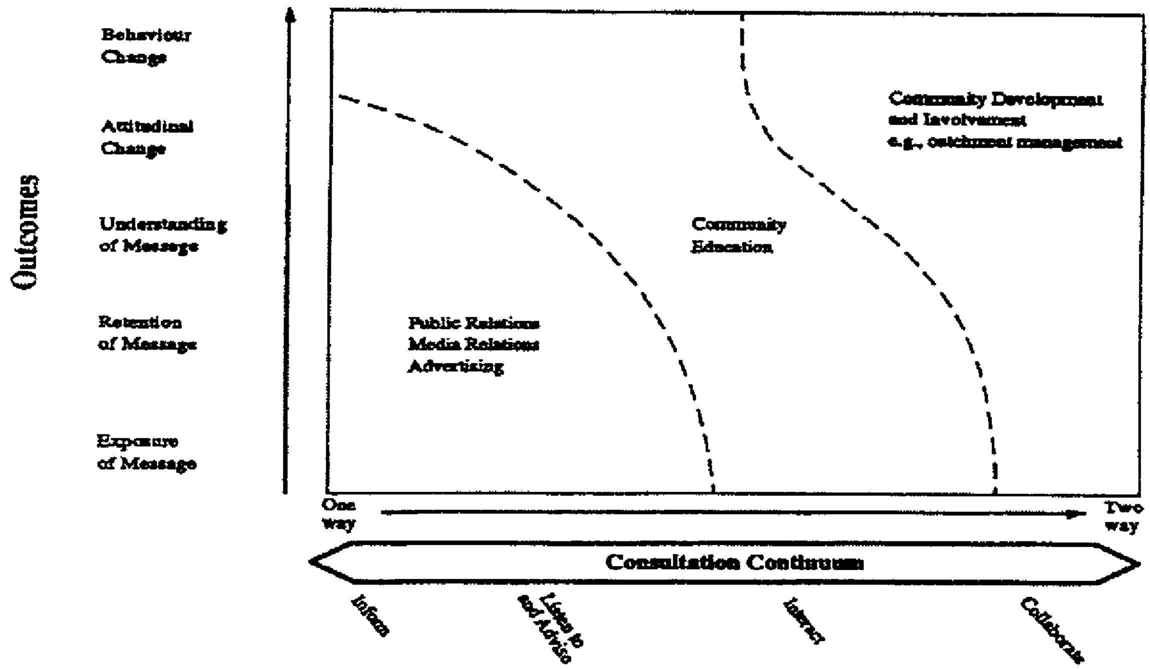
At a community level, a long-term indicator of interest in reuse will be willingness to invest in water savings measures - the benchmark cost for renovating waste water is currently around $1.50 per kilolitre versus water which currently retails in Canberra at around $0.35 per kilolitre ($0.80/kl above 350 kilolitres) and as yet unresolved the costs of using other water sources -groundwater, storm water, lake water and riparian use of streams. Such costs beg the question of the environmental value we all place on high quality water resources such as those currently preserved in perpetuity for water supply.

One useful model of community participation is reflected in the National Water Quality Management Strategy *Implementation guidelines*. The diagram, which shows the importance of taking time and making some prodigious effort to communicate the issues, without prearranging outcomes, speaks for itself.

Environmental and water availability imperatives will dictate that those who want water will have to pay a premium for using the resource. The issue of water supply was visited at length in ACTEW's *Water supply futures strategy*. That strategy identified reuse as one savings option to head off increased potable demand and any short term need for a new dam.

Whether savings levels occasioned by reuse will become very significant, say reach as much as say 40% of current potable supply, remains to be seen. Limits to reuse also need to be considered in the context of environmental return flows of renovated wastewater such as those identified in the *Environmental and Process Audit* of the Lower Molonglo Water Quality Control Centre.

Communication Model



Source: National Water Quality Management Strategy Implementation Guidelines.

REUSE AND THE QUANTITY FACTOR

A truly sustainable model of water use would have us *all* extracting minimal water, maximising its use and returning renovated water of a high quality back to receiving waters wherever possible. Reuse offers the potential to limit disturbance to the hydrological cycle by limiting abstractions, and at least delaying future pressures on our water resources. Such pressures may grow in a less and less predictable manner with sudden changes to population or with the arrival of water demanding industries. One paper pulp mill to process pine could change the water balance equation overnight!

A World Health Organization imperative of nearly forty years ago stated that *No higher quality water, unless there is a surplus of it, should be used for a purpose that can tolerate a lower grade.* While the context was, no doubt, one of preserving and protecting potable water for human needs, there is now a clear recognition that we have equally to protect the wider water environment if, for no other reason, in the great hope that we can, along with the environment, survive and prosper.

The most seminal reason to address the water conservation issue relates to *sustainability*. Issues such as environmental flows, or minimum perturbation to the hydrological cycle, must be addressed in any considered appraisal of the value or importance of reuse. Reuse is neither a panacea for limited water resource problems nor for problems associated with the urban water cycle. Reuse is sometimes painted as the solution to problems such surface water eutrophication problems. Nevertheless reuse must be provided every opportunity to contribute to water resource management.

WHAT REUSE HAS TO OFFER THE COMMUNITY

One positive opportunity for reuse is that it will bring a new order of reality and technical innovation to the water industry. The water industry has built its dams, its sewerage systems and its water supply reticulation infrastructure. For many years it has been undergoing radical reform, first augmentation and management of its systems in the 1970's and 1980's and more recently radical reform to its business infrastructure. Reuse provides opportunity for technical innovation tempered by community desires to use water sensibility and economic feasibility. Some of these opportunities may see expression in:

- a more rational approach to valuing of water and wearing the costs of water renovation;
- sewerless suburbs with trickle feed potable supply to buffer rainwater tanks; and
- a rewatered landscape where potable water only meets internal domestic requirements.

A broad change to urban water use practices is inevitable and will place demands on the community and agencies. One opportunity is to partner reuse opportunities as already alluded to. However, until the value of potable water resources is recognised through more realistic pricing mechanisms (Europeans typically pay five to ten times Australian municipal prices in a water abundant environment), we see public enthusiasm for reuse being inhibited by commercial questioning of its viability. This has to change before we enter the phase of water wars, the beginnings of which are so evident from the irrigation lobby challenging capping extractions from the Murray Darling Basin.

WHERE DOES REUSE SIT IN THE WIDER CONTEXT OF CORPORATE RESPONSIBILITY?

ACTEW Corporation and the EMIAA have been on much the same learning curve as the outside world in relation to wastewater reuse. Each has had its own idiosyncratic corporate vision of how the community might use its water resources more sustainably. We certainly aware of water resource issues - that, from a use point of view, is our business. As with a locally conducted *Water supply futures strategy*[1], we need to *foresight* responsible water management initiatives, of which water supply demand and reuse programs are two more obvious options.

We are asking the community what else we can do with wastewater, how to limit its generation in the first instance, and how to better view it as a resource rather than as a waste stream to be dealt with by society. That old-fashioned notion of stewardship of our resources again comes to the fore. If the waste stream is renovated, it can be returned after people have *borrowed it and returned it in good condition from whence it came,* Our vision is that we go one step further, consider the option of reuse and consider the potential for reuse to limit potable water abstraction and optimise the value of water to the community.

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About the authors

Paul J Perkins is the Acting Chief Executive Officer at ACTEW Corporation and has been at the forefront some of its major consultations including a water audit of its Lower Molonglo Water Quality Control Centre.

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HISTORICAL DEVELOPMENT OF WATER POLLUTION CONTROL AND UPDATE TOPICS OF SEWAGE WORKS IN JAPAN

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HISTORY OF WATER POLLUTION CONTROL IN JAPAN [1]

Prehistory of Modem Sewage Systems

Water pollution has been an issue ever since people began to live in cities or areas of high population density. The first signs of water pollution took the form of hygiene problems resulting from the pollution of drinking water in industrialized and urban societies in European countries in the early 1830s. Although knowledge about pathogenic bacteria was inadequate at that time, people began to build large-scale sewer systems and install flush toilets in order to remove human waste from the living environment as quickly as possible.

The direct discharge of domestic wastewater through the sewer systems into rivers and the waters into which they flowed caused water pollution in rivers and natural water bodies. Since water pollution was caused mainly by organic loading from domestic wastewater, priority was given in this period to the development of methods to remove organic substances from raw wastewater before it was discharged into rivers.

Various biological wastewater treatment methods have been developed and implemented since the middle of last century with the aim of removing organic materials. The original activated sludge process was invented in 1914 in Manchester, England. Various modifications of the process have been invented and applied since then.

The people of Japan have historically been highly aware of sanitation issues, and strict systems for the management of human waste, such as night soil, and the maintenance of safe water supplies in urban areas were developed early in Japan's history. These factors are reflected in low death rates from water-borne diseases, and in the rather slow development of sewer systems. However, the rapid growth of urban populations and the development of heavy and chemical industries led to serious water pollution in rivers and sea areas, and serious pollution problems in major cities. and industrial areas.

Water Pollution Control in Japan

Economic growth since the mid-1950s has been accompanied by an increase in both the qualitative and quantitative environmental impacts of wastewater, especially from heavy and petrochemical industries. Tragically, some serious cases of environmental pollution were manifested through their impacts on human victims.

One such case was the outbreak of Minamata disease, which in 1956 was officially attributed to methyl mercury in wastewater. By 1989 there were 1,755 certified victims of Minamata disease, but many more sufferers are seeking official certification under the Compensation Law. Itai-itai disease, which was caused by cadmium in wastewater from a zinc mining operation, was identified in 1958. In 1968 PCB contamination in rice oil caused PCB toxicosis. These examples are indicative of various forms of environmental pollution of the water and direct contamination of food itself.

The industrial wastewater pollution in the late 1950s became a very significant and serious issue in relation to water pollution control policies and technologies. It was found at this time that trace amounts of toxic heavy metals could be accumulated in living things, including fish, shellfish and crops, and that these toxic substances could have a direct impact on human health when ingested through contaminated foods. These toxic substances are never degraded biologically during treatment processes or in the natural environment. It is therefore extremely important to restrict the discharge of these toxic substances in the environment even through sewer systems, and to control them at source through the prohibition of their usage or production.

Fig.1 The Sumida River in the Early .1960s

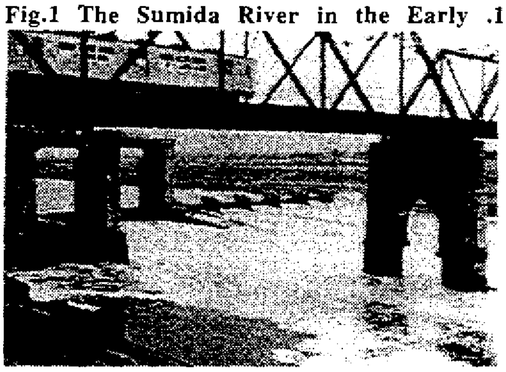


Fig.2 The Tama River in the Early 1960s

Figs. 1 and 2 show water pollution in the Sumida and Tama Rivers, which are two largest rivers in the Tokyo Metropolitan Area, in the late 1950s to early 1960s. Fig. 1 shows large amounts of floating material. It was reported that the BOD of water in the river at that time was around 30mg/l. Fireworks displays and college regattas were suspended in 1961 and did not resume for about 15 years. Fig. 2 shows large quantities of foam generated by the hard-type detergent, ABS, on the surface of the Tama River.

These tragic and heavy experiences of environmental pollution led to the drafting of the Environmental Pollution Prevention Law (Japan's basic environmental law), which was first enacted in 1967 with the aim of preventing environmental pollution in general. However, efforts to overcome environmental pollution in Japan ware not notably effective, and environmental protection became one of the biggest areas of public concern. In 1970 the basic environmental law was revised and strengthened during what became known as the "antipollution session" of the Japanese National Diet. The Diet also enacted 14 related laws dealing with various aspects of environmental pollution, including air pollution, water pollution, marine pollution, solid waste management, sewer construction and so on.

Table I Environments and Laws in Japan

Environments	Laws
Atmosphere	Air Pollution Control Law Offensive Odor Control Law
Water (Rivers, Lakes, and Inland and Coastal Seas)	Water Pollution Control Law Sewarage Law
Ocean	Law for the Prevention of Marine Pollution and Maritime Disaster
Land (Agricultural use and Landfill site)	Agricultural Land and Soil Pollution Prevention Law Waste Disposal and Public Cleansing Law

Seven laws stipulate responsibility for the final discharge and disposal of waste in specific types of environments. Environments for the final disposal of waste are classified into the atmosphere, water, the oceans and land. The seven laws corresponding to these four categories are listed in Table 1.

The environmental standards for the ambient air and water bodies are determined by the Environmental Pollution Prevention Law. Emission standards for flue gases are regulated under the Air Pollution Control Law, while the Water Pollution Control Law stipulates effluent standards for wastewater. (Details of parameters and standards are not cited here.) Verification standards set down in the Waste Disposal and Public Cleansing Law are used to specify hazardous wastes in relation to final disposal of solid wastes. The verification standards cover concentrations of hazardous substances in wastewater as determined through elusion tests of solid wastes, and amounts of hazardous substances in solid wastes. The elusion test standards are similar to the effluent standards for wastewater.

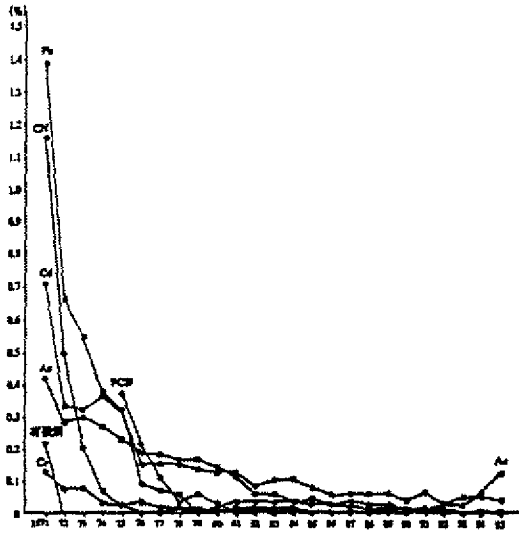


Fig.3 Changes in the Non-compliance Ratio of Environmental Water Quality Standard for Toxic Substances (mercury has not been detected for these period)

Yearly changes in non-compliance ratios for environmental standards for toxic substances in relation to human health were shown in Fig. 3. It is clear from the graph that industry efforts, public support and government regulation have been considerably more successful in improving water quality than regulation relating to ordinary organic substances as shown in Fig. 4.

Fig. 4 traces trends in organic water pollution in terms of progress toward the attainment of environmental quality standards, as represented by in rivers and COD (Mn) in takes and sea areas. Generally speaking, the restoration of water quality in natural water bodies is a slow process, and much effort is still needed to improve the water quality situation in Japan.

In 1972 serious damage to fisheries was reported as a result of a "red tide" in the Seto Inland Sea. After this incident, the problem of eutrophication in both closed sea areas and lakes reservoirs became a key area of concern about water pollution. Serious consideration was given .to various approaches to the minimization of the amounts of nutrients discharged into natural waters. Among the methods studied were the development of phosphate-free detergents, the introduction of nutrient standards for effluent and environmental standards, and the application of nutrient removal processes as an advanced treat-ment in conventional secondary treatment facilities. Fig.5 shows yearly changes in the incidence of "red tides" and actual damage to fisheries in the Seto Inland Sea area.

Although the graph shows that the incident of " red tide" outbreaks in the Seto Inland Sea is decreasing, eutrophication in lakes and reservoirs and other bays and sea areas is still the focus of considerable concern and has become a crucial water pollution issue in Japan. As shown in Fig. 4, COD (Mn) levels in lakes and sea areas do not indicate any clear improvement in attainment ratios for environmental standards. The slow pace of improvement is thought to result in part. from delays in adding nutrient removal processes to treatment systems for both industrial and domestic wastewater.

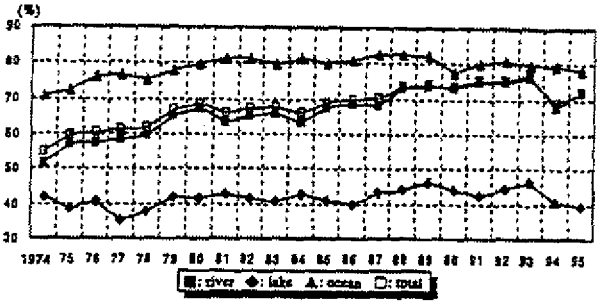


Fig.4 Changes in the Compliance Ratio of Environmental Standard of Organic Parameters (BOD far rivers and COD(Mn) for lakes and sea areas)

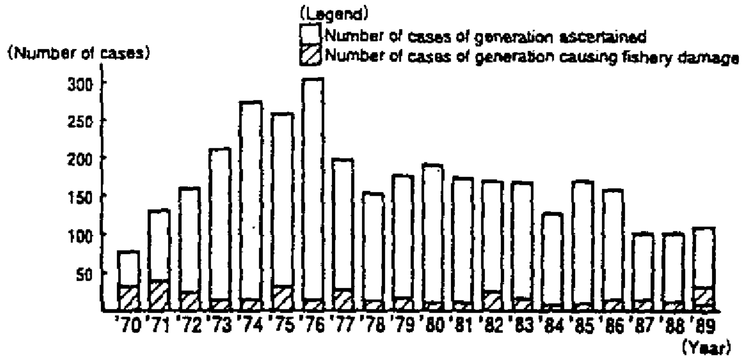


Fig.5 Trends in Number of Ascertained Generation of Red Tide in Seto Inland Sea (Surveyed by Fishery Agency)

Other recent topical issues relating to water pollution in Japan include the safety of drinking water and the problem of global warming. The safety of drinking water from the pathogenic bacteria, like 0157, and the pathogenic microorganisms, like Kriptpsporisium, has been discussed again as a very urgent issue in the water supply field, although it was the first issue in the history of water pollution in early stage of development of sewage works in the world. The method of disinfection is now investigated to ensure the disinfection of water and no harmful byproducts from disinfection processes.

In addition to this new concern about the safety of drinking water, global warming problem has also emerged as a serious problem for future generations. The relationship between water pollution and global warming is rather complex, since some wastewater treatment processes may produce greenhouse gases, such as methane (CH_4) and nitrous oxide (N_2O), while complex ecosystems in treatment systems and natural water environments could be affected by global warming. To prevent the excess release of methane gas to the atmospheric environment it is very important to use the methane gas as effective energy source. Regarding the production of nitrous oxide from biological nitrogen removal processes, the basic investigations have just started and details of mechanisms of N_2O production should be made clear further more.

A chronological table of major environmental issues and the substances that have been regarded as major parameters in water pollution problems in each era are summarized in Table 2.

Table 2 Historical Changes of Hazardous Substances in Water Pollution

Era	Hazardous Substances	Countermeasures
1800-	Pathogenic bacteria	Construction of sewer system and disinfection
1900-	Organic materials (BOD material)	Biological treatment and physico-chemical treatment
1950-	Heavy metals and non. biodegradable chemicals	No discharge to environment (Treatment at source)
1970-	Nutrients for eutrophication	Nitrogen and phosphorus removal processes
1980-	Trace substances (Carcinogen, off-flavors and taste)	Activated carbon or membrane technologies
1990-	Green house gasses (CH_4, N_2O and CO_2)	Energy saving technologies

UPDATE TOPICS OF DEVELOPMENT OF SEWAGE WORKS IN JAPAN

Development of Construction of Sewerage Systems

Long and intensive efforts to develop sewer systems and treatment facilities, the total replacement of hard-type synthetic detergents with soft-type biodegradable products, and the strict application of effluent standards for industrial activity have brought water quality in the major rivers back to good water quality condition. For instance, the considerable improvement of water quality of the Sumida river has been confirmed by the facts of low BOD concentration and fish returning the river. In the Sumida river, a variety of river activities, including fireworks displays, college regattas and sight-seeing boat trips, have reopened there for people's enjoyments of water environment.

In the one of the tributaries of the Sumida, the Kanda river, the typical example of relationship between the river water quality recovery and the expansion of the sewerage systems is shown in Fig. 6. Although about 95 percent of the flow rate of the Kanda river is come from the effluent of one big sewage treatment plant in Tokyo, it is clearly shown that the river water quality has been improved dramatically depending on the increase of the percentage coverage of sewerage systems and introduction of advanced wastewater treatment facilities in the treatment plant.

The present general situation of the construction of sewerage systems in J Japan is summarized in Fig. 7. In the eleven Major cities of which population is larger than one million, the percent population served by public sewerage systems is come over 97 percent in 1995. Thirty percent of the total Japanese population Eve in 2,791 small cities of which population is less than 50 thousand. In these cities, however, only 17 percent of the population is served by public sewerage systems. Many small cities are located upstream of important fresh water resources. To protect the water quality in rural areas, so large number of small size sewage treatment plants should be constructed very soon. The treatment plants there should be developed new way to meet the limited financial condition of small cities for construction, operation and maintenance costs of tile sewerage systems.

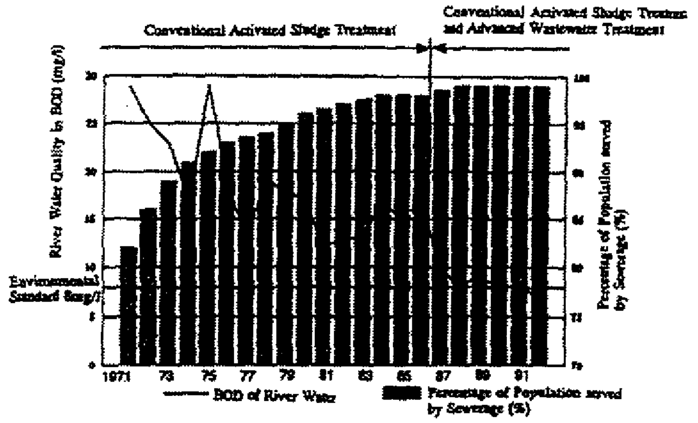
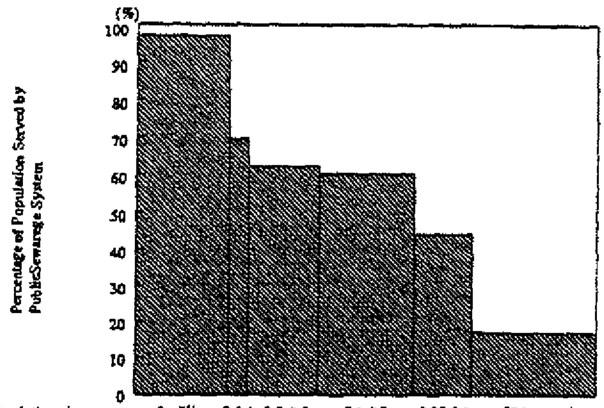


Fig.6 River Water Quality and Development of Sewerage Systems and Treatment in the Kanda River in Tokyo'



Population size	>1million	0.5-1	0.3-0.5	0.1-0.3	0.05-0.1	<50thousand
Total population	25millions	6	17.4	25	15.6	35.5
Served Population	23millions	3.8	10	13.2	6	4.6
Number of Cities	11	9	45	153	227	2,791
Sewered Cities	11	9	45	150	199	670

Fig.7 Construction of sewered systems in Japan

The average percentage of population served by public sewerage systems is still 54 percent in Japan. The primarily important issue of national government of sewage works is to how expand sewerage systems for all over the country. But at the same time, the major large cities, where the percentage itself have come to close to 100 percent, have to face severely to various problems of new requirements such as enhancement of nutrient removal efficiency to prevent the eutrophication in down stream lakes and closed sea areas, need of advanced wastewater treatments for introduction of reuse systems of sewage, and development of sludge treatment and disposal systems to meet the scarcity of sludge dumping sites.

Framework for small scale wastewater collection and treatment systems in Japan

As mentioned in the previous section, the most serious current concern in the expansion of public sewage system is how to expand its coverage into small scale communities. Generally speaking, those rural communities have low population density and residents locate scattering around, so the cost of collecting sewage comes to be very expensive compared with that of the large cities. In the collection system of sewage, besides conventional gravity-flow collection systems, vacuum sewers or pressure sewers systems are considered as a possible less expensive collection systems for small communities.

Also in the operation and maintenance practice of the treatment plant, the sludge handling process is thought to be a most troublesome and expensive process in the small scale facilities and it is a recommendation of selecting a less sludge yielding biological processes such as oxidation ditch and extended aeration system. At the same time, because of the simplicity of operation, sequencing batch reactor, aerobic filter, contact oxidation, and rotating disk systems are recommended as usable type of reactors in the small scale facilities. And a centralized monitoring and management system is also planned for the daily operation and maintenance for some plants.

For the sludge treatment, it is some time planned to introduce a partly centralized system in which a car with storage tank and vacuum pump rotates around some small facilities to collect their sludge and the collected sludge is treated at a one of the facilities of them. In other case, a car equipped with a dewatering and drying facilities of sludge rotates to small scale treatment plants for the on site sludge treatment, and the dried sludge is composted and used for agricultural purposes.

The Case Study of Advanced Wastewater Treatment in the Lake Biwa[2]

The Lake Biwa is the largest lake in Japan and provides drinking water for 14 million people of Kyoto and Osaka regions. The water quality of the lake started deteriorating 30 years ago mainly due to economical development in the watershed of the lake. Water bloom caused by algae were frequently observed and odor and taste problems in tap water occurred in summer time. TO protect and restore the water quality, introduction of an advanced wastewater treatment process was planned in 1971.

The Konan Chubu plant, the largest plant there, has a capacity of 111thousands m^3/day, equipped with a single sludge type pre-denitrification process with aluminum polymer addition followed by rapid filtration. Fig. 8 shows a flow scheme of the Konan Chubu plant. In 1993 the Konan Chubu plant received an influent flow rate of 78 thousand m^3/day and effluent quality was much better than the operational target as shown in Table 3.

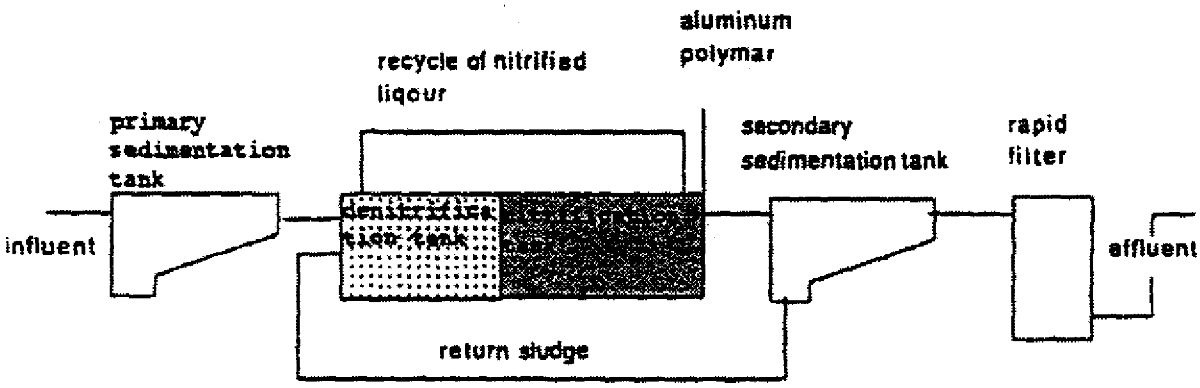


Fig. 8 Flow Scheme of Konan Chubu plant

Table 3 Water quality of influent, effluent and operational target at Konan Chubu plant

	BOD	SS	T-N	T-P
Influent	181	178	30.5	3.38
Effluent	n.d.	0.6	7.0	0.06
Operational target	5	6	10	0.5
				unit mg/l

Annual average in 1993

Wastewater Reuse Systems

The tightness of water resources in large cities like Tokyo and Fukuoka, Japan, has come very severe because of the increase of water consumption for urban activities. For the purpose of the water usage like toilet flush' cooling and air-conditioning, sprinkler, washing and water environmental restoration et al., ' it is not required to keep the cleanest water quality just for drinking water. Wastewater reuse systems have been developed first mainly for industrial usages like cooling and air-conditioning, washing, and gradually developed various purposes like toilet flush, sprinkler, and water environment restoration.

A large scale regional sewage reuse system has started practically since 1985, after the severe experiences in water shortage in 1978. Adding to the water shortage problem, a growing interest in energy conservation caused by the oil shock in 1973 was favored the introduction of wastewater reuse systems by the national and local governments. In recent years, the systems have been introduced at the average rate of nearly 130 per year as shown in Fig. 9. As of 1993, treated wastewater was being used at 1,963 facilities nationwide. And the purposes of the reuse systems are classified as shown in Fig. 10. The most common use is for the toilet flush use in the building. In terms of the amount of water used, approximately 277,000 m^3 of water are estimated to be reused per day throughout the country, which is equivalent to roughly 0.7% of the amount of domestic water used in Japan.

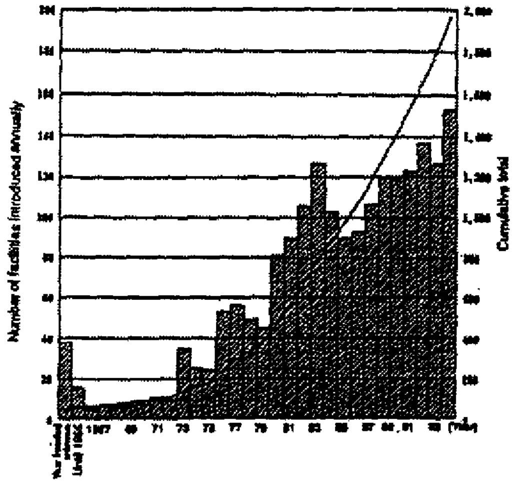


Fig.9 Number of Facilities of Wastewater Reuse Systems (National Land Agency in 1993)

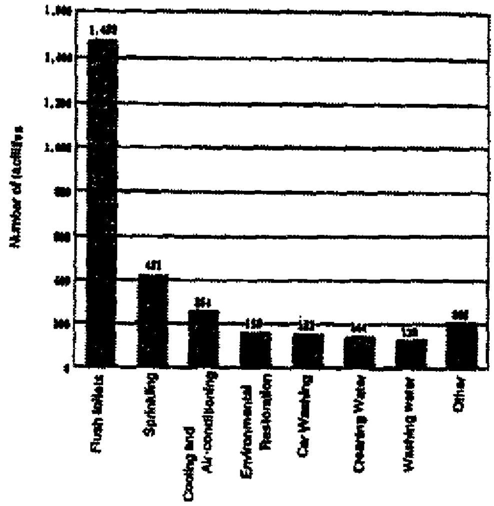


Fig.10 Classification of Facilities of Wastewater Reuse Systems (National Land Agency in 1993)

Another Aspects of Development of Sewage Works

1) Heat Recovery from Sewage

In Japan recently, people has started to concern very much about the global warming problems and energy savings in urban activities. It is thought that the sewerage system is a good and effective facility to collect and recover the wasted heat from people's usual activities. The temperature of the sewage is generally much **warmer than ambient air temperature in winter** and much cooler in summer. The heat pump system is used for room heating in winter and cooling in summer from aspect of saving the electric power consumption and usually the open air is used as a hot heat source and a cool heat source for room heating and cooling, respectively. The total amount of possible heat recovery is estimated as 9,000 billion kcal/year, it is about 3.7 billion kw in electric power and also about 0.8 billion m3 of city gas equivalent, in Tokyo Metropolitan area for the case of 5 °C difference in water temperature and air temperature.

2) Optical Fiber in the Sewerage System[3])

To exploit the full potential of information technology - particularly in a city of 1.2 million, as is the case for Tokyo Metropolitan area - it is fiber optic systems which are seen as the means by which the huge volumes of information traffic can be transmitted. The problem is how to install such systems.

Installing fiber optics above ground is seen as easiest and cheapest but this option is increasingly regarded as a form of visual pollution. The alternative, to install such systems underground, may be more expensive but one significant point in its favor is that, once installed, it is more secure. For example, the Hanshin Earthquake that struck Kobe city in 1995, killing more than 6,000 people, indicated that the telephone cable under the ground was 80 times safer than that above ground.

Problems of installing cable underground has to compound numerous infrastructures which have already installed in the underground, such as water works, sewerage, subway, electric power, gas and telephone. Tokyo Metropolitan Government determined to exploit information technology to the full scale and set out in the mid-1980s, through its Bureau of Sewerage (BOS), to investigate the potential of installing a cable system in its sewer network. Though the previous low of sewage works in Japan did not allow to use the pipelines for any other purposes than conveying sewage, the low was amended in May 1995 to allow the installation of optical fiber in side the pipes so far they do not give substantial damage to the original function of the sewerage system.

Now the BOS of Tokyo Metropolitan Government has completed the optical fiber cables with a total length of about 214 km as of March 1996. The BOS has been working on a new project of the SOFT Plan, Sewer Optical Fiber Teleway Network Plan. In the SOFT Plan, various experimental menus as Multimedia information service, Regional public information service, Remote health consultation service, and Remote education service, will be investigated.

Regarding the technical issues of the practical application of the optical fiber in the sewerage pipes, it is said that the fiber optic cable system has to satisfy five conditions for it to be suitable for installation in sewers; it should be able to withstand contact with wastewater, gas and acid; it should be able to withstand the water pressures (of 150 kg/cm^2) used during sewer cleansing; it should be possible to make the necessary connections in the space available; and the system should not reduce the capacity of the sewer.

CONCLUSION

Japanese sewage works has developed gradually from the stage of construction of sewerage system for collecting the sewage and introduction of conventional treatment facilities into the stage of the introduction of advanced treatment for nutrient removal and reuse of wastewater, and further into the additional usage of the heat of sewage and sewer pipes as indirect use of sewerage systems. Although the details of sludge treatment and disposal problems are not discussed in this paper, the sludge issues arc very urgent one for a concept of holistic approaches to the wastewater management from both of global and local environment preservation.

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THE CONCEPTION AND EXECUTION ON THE IMPROVEMENT OF RIVER WATER QUALITY BY A NEWLY DEVELOPED PURIFICATION METHOD: " SHIMANTO-GAWA SYSTEM "

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THE BACKGROUND OF THE SYSTEM

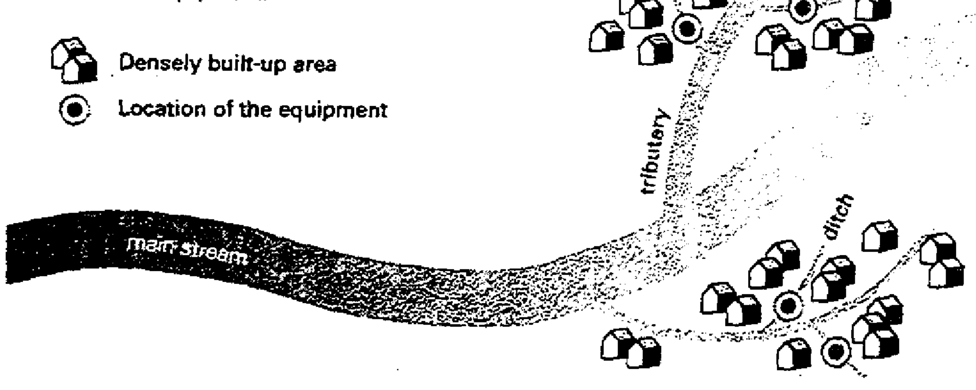
The former Japanese nature was praised as a symbol of a famous country for its natural beauty from many foreigners. With high growth of the Japanese economy, however, many people have kept away from the nature and become indifferent to it. As a result of the lack of consideration to the nature, various kinds of pollution has gradually invaded in places and the nature has lost its beauty with rapid progress. Especially water pollution in hydrosphere has developed seriously not only in urban areas but also in rural ones owing to changes in their ways of thinkings and in their life-styles which had been never taken root into the climate of Japan. Moreover, a vast quantities of imported food and feed materials from overseas countries has already been beyond the environmental capacity of a cycle of these substances in Japan. Most of Japanese rivers and lakes, therefore, are now suffered from various kinds of pollution including eutrophication, high BOD (biological oxygen demand) and synthetic detergents because that there are very few treatment plants with more higher grade treatments, for example, nitrogen removal facilities from the secondary sewage effluent.

This paper deals with the technological approach to improve the water quality of rivers and lakes in Japan by using only natural materials without any synthetic chemical ones for its purification.

THE CONCEPT OF THE NEWLY DEVELOPED EQUIPMENT FOR THE IMPROVEMENT OF RIVER WATER QUALITY

It is needed for the equipment to treat a small quantities of discharge at the small outlet of foul water or from the small drain which is running as a stream since it will become too huge volume of water to treat for purification if the foul water pour into the main course of a river. Therefore, the installation of the equipment is proper such as small ditches where can be settled directly at the bottom of the stream or just under drainpipes collected foul water from small drains. Furthermore, when it rains heavily and quantity of drainage increases the concentration of pollution substances in original drains will be diluted by rain water. In such a case, the inlet gate of the equipment is required to shut automatically or to control constantly the volume of drainage water in order to protect functions for purification inside of the equipment from the inflow or earth and sand. Fig. 1 shows the conceptualized locations of the equipments along the explanation mentioned above. To treat nutrient salts such as nitrogen and phosphorous in sewage effluent into the ultra-low concentration will

Figure 1: *The conceptualized locations of the equipment.*



be generally required the very high-priced facilities. For this reason, almost of all local governments in Japan cannot install the treatment station with higher grade removal systems for nutrient salts.

It being needed to develop the equipment to reduce the concentration of nutrient salts at low cost performance as well as BOD, COD, SS (suspended substances), heavy metals, and synthetic detergents, my laboratory colleagues and I started to develop the equipment filled up with only natural materials for the following purposes (1) to make sure of microbial habitation for the promotion of organic decomposition and biochemical reactions, (2) to absorb inorganic ions dissolved in the treated water and (3) to make more easier and safely restore saturated absorbents to soil for working natural circulatory system in earth.

For the reduction of nitrogen concentration in sewage effluent or foul water, the mechanism of nitrogen starvation in soils is applied into practice. Namely, when organic materials with high carbon-nitrogen ratio are buried in soils microorganisms in the soil are apt to use the nitrogen in soil to keep balance of their body's component, so that plant growing on the soil fall into nitrogen deficiency. We collected fallen leaves, withered branches and half-decomposed trunks from a lushly forested land and put them into a nettled nylon bag. This is one of materials which are plugged in a coupled concrete chambers described later. Almost of all heavy metals are expected to precipitate as sulfides in a reduced condition which is generated in an anaerobic liquid phase at the first and large chamber of the equipment. Phosphorus ions will be absorbed at the actively absorbing sites on a processed lime stone to promote the absorbing ability for phosphorus. A named "bio-charcoal" is designed to make sure of microbial habitation in a mobile liquid phase. For this purpose, a special kind of charcoal with many and strong fine porous were produced and it was treated with crude chitosan which are obtained from crab shell. By microscopic observation it was confirmed that many fine tubes of the charcoal coated with crude chitosan can provide habitations for microorganisms to propagate by gaining their shelters from attacks of protozoa while the untreated charcoal is kept from the settlement of microorganisms (Photo. 1). Another microscopic observation on the behavior of bacteria in an untreated charcoal suggested that many fine carbon projections like needles formed in tubes of the untreated charcoal will disturb the deeper entrance of microorganisms into the tube. Functional purification by bio-charcoal

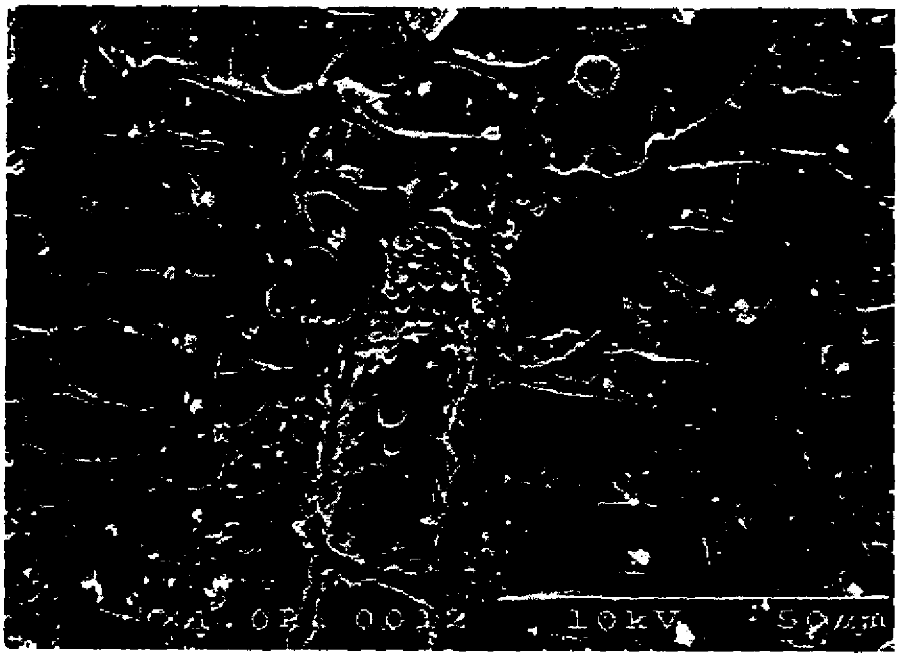


Photo 1: Microbial habitation in tubes of charcoal treated with chitosan.

with microbial habitation will be expected to promote the decomposition of various kinds of organic matters which are not only easily decomposable organic matters but also slightly decomposable ones like high polymer organic compounds.

REALIZATION OF THE PURIFICATION SYSTEM : "SHIMANTO-GAWA SYSTEM "

" Shimanto-gawa " is the name of the river which has its full length 196km and streams in the western parts of Kouchi Prefecture, Shikoku District, Japan. There are two reasons why the purification system was named particularly after Shimanto-gawa although Shimanto-gawa is neither the longest river nor the largest basin in Japan. One reason comes from that Shimanto-gawa is very famous for its natural beauty in Japan, and has been called and loved by the last limpid stream from many Japanese. The other reason is that this system has been developed and realized by the technical committee on natural circulation system for the river water treatment which was organized by specialists and officers from University of Tokyo, Kouchi Prefectural Government, Ministry of Agriculture, Forestry and Fisheries of Japan, some municipalities of Shimanto-gawa basin and Toyo Denka Industry Co., Ltd.

Fig. 2 shows the cross section of the basal structure of Shimanto-gawa System. The system is composed of five serial chambers and each chamber is connected by opened channels. The first chamber of the five serial ones is called by the precipitation tank to cause not only earth and sand, but also heavy metals to precipitate as sulfides under the reduced condition. The second tank after precipitation one is charged with filters to make suspended substances in the stream adhere and with fallen leaves, withered branches and

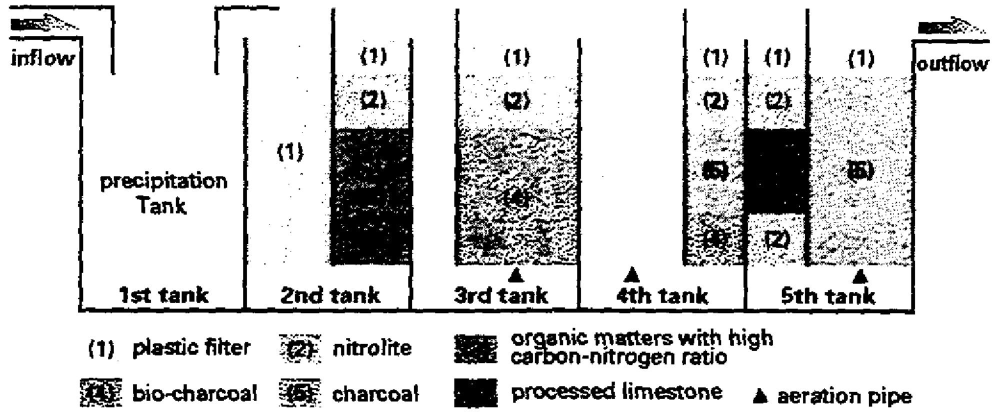


***Figure 2:** The cross section of the basal structure of Shimanto-gawa system.*

half-decomposed trunks to eliminate the inorganic nitrogen compound dissolved in the stream such as nitrates by the biological reaction of microorganisms as mentioned above. The named "nitrolite" plugged in this tank is a some processed zeolite for absorbing ammonium ions. The third and forth tanks are plugged with bio-charcoal, nitrolite, charcoal and filters using in the second tank. These tanks are expected to give full scope to their abilities for the reduction of concentration of slightly decomposable organic matters such as synthetic detergents and to promote the decomposition of macromolecular compounds such as carbohydrates. And the fifth tank is charged with nitrolite, charcoal and a some processed lime stones for absorbing phosphorus ions. Depending on the condition of amounts of water to treat, water quality and geographical features of the settlement place of the equipment the structural material, scale and contents of the equipment will be modified. The equipment will be constructed on the slightly gradient basement (1/80~1/100) to hasten the natural stream smoothly and running without pump power. The aeration pipes are installed at the bottom of the third and fourth chamber and supply air intermittently. If abundant sunshine is available, the electricity from solar cells will be able to provide enough energy to allow motors to run for the aeration. The first equipment with treatment capacity of $70m^3$ /day has been set up since March 1993 at the small outlet of ditch in the village along the basin of Shimanto-gawa. Up to the present of the end of November, 1996, 17 Shimanto-gawa systems including the largest facilities with treatment capacity of $2{,}400m^3$ /day have been built in places in Japan and all of them have been satisfactorily running without any special repairs. Photo. 2 shows one of the comparative sceneries at the before and after the installation of equipment.

ANALYTICAL DATA OF WATER QUALITY OBTAINED FROM EQUIPMENTS

Periodical data on chemical analyses and a bacteriological examination of water sampled from each chamber in equipments have been collected from the first one month after the initiation of the operation. At the same time several kinds of observation on not only the change of water quality but also ecological changes of aquatic lives at the outlet of the installation have been reported.

1) Chemical analyses

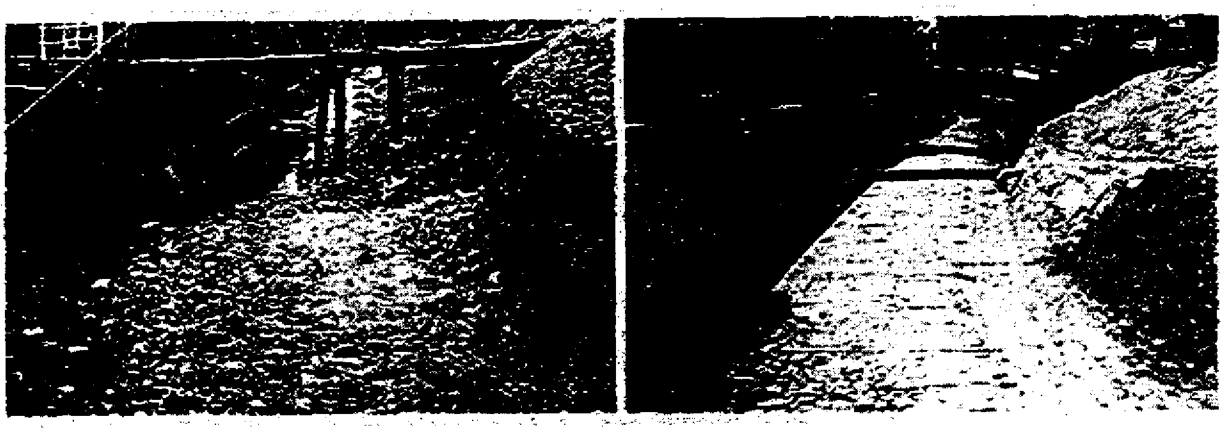


Photo 2: Comparative sceneries before (left) and after (right) the installation of equipment.

	BOD		COD		Total Nitrogen		Total Phosphorus		Detergents	
	Inlet	Outlet	Inlet	Outlet	Inlet	Outlet	Inlet	Outlet	Inlet	Outlet
average (mg/l)	43.0	3.6	26.0	4.7	1.80	0.77	0.77	0.19	0.660	0.057
maximum (mg/l)	330.0	6.9	220.0	7.6	6.50	1.20	1.20	0.94	3.100	0.170
minimum (mg/l)	1.6	0.7	1.4	1.8	0.32	0.23	0.23	0.04	<0.001	<0.001

Table 1: *Water quality at the outlet and the inlet of the equipment.*

Table 1 shows the average, maximum and minimum values of water qualities sampled during a whole day at the inlet and outlet of the equipment installed in 1993. From the table the elimination percentages of BOD, COD, total nitrogen, total phosphorus and total detergents are calculated as 93, 82, 61, 66 and 92% respectively. The tendency is that comparing with elimination rates of BOD, COD and total detergents, those of nutrients salts are not so high is generally recognized in every equipment. The reason why elimination rates of nutrients salts in the stream cannot be raised to more than 80% will be based on the fact that total nitrogen concentration in streams in Japan being seldom over 10ppm the incorporation of nitrogen to microorganisms is not so high as the case of soils, and phosphorus concentration in streams being also seldom over 3ppm the absorption of phosphorus is not so active by the processed limestone. In spite of the low concentration of detergents their elimination rate is very high. This will be explained that judging from the isothermal absorption diagrams between various kinds of detergents and bio-charcoal. Fig. 3 gives changes of purification abilities in an equipment as the years go on about BOD, total nitrogen, total phosphorus and total detergents. The purification ability of the equipment has been kept in constant and well maintained for more than two and a half years in spite of there being low temperature of stream in winter seasons. We hope that the

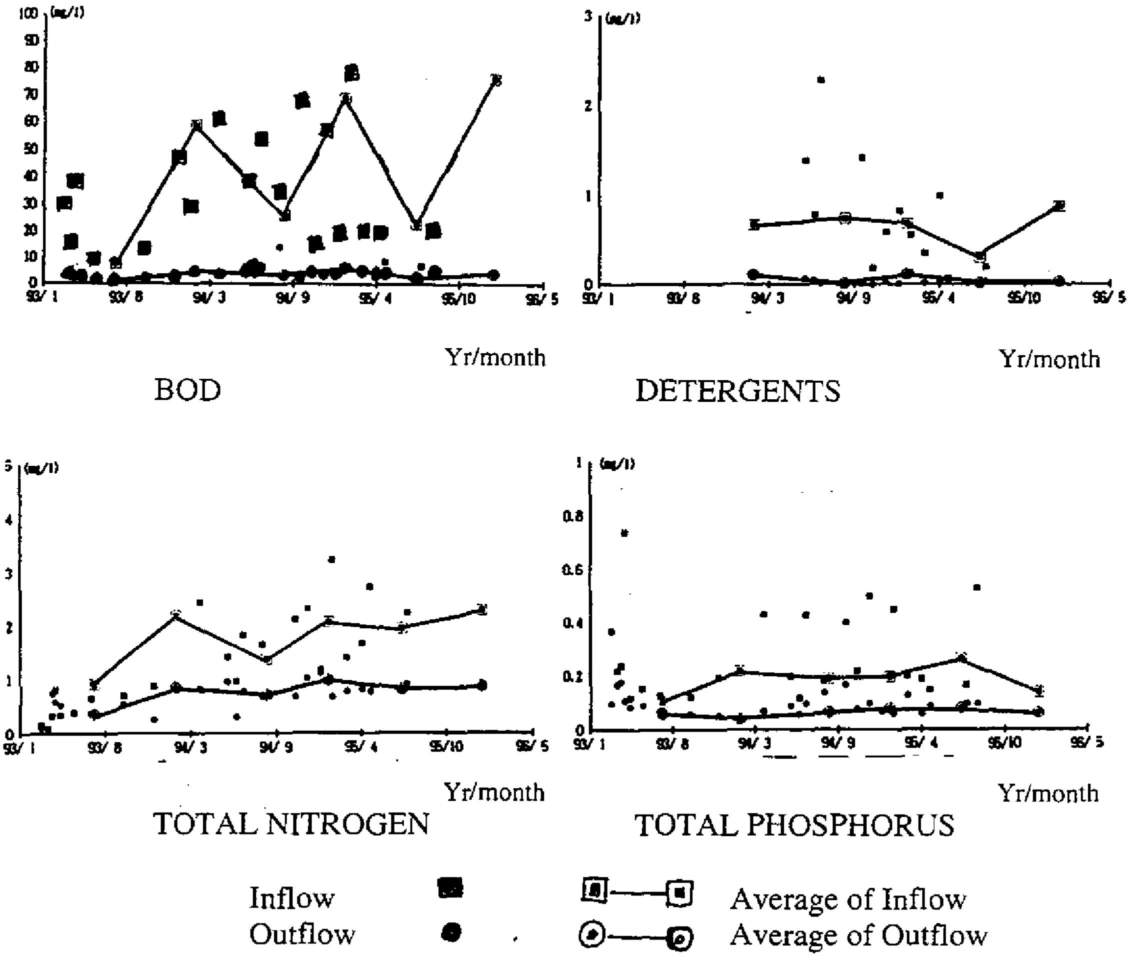


Fig. 3 Changes of purification ability in equipment as the years go on.

equipment will brings its ability into full play for more than 10 years without any exchange of plugged materials in each chamber.

2) Bacteriological examination

Table 2 shows the detection results of bacteria counts at the inlet and outlet of the equipment in the summer season. The equipment also gives us a good suggestion about the elimination of bacteria count in the stream and the possibility of a comeback for the healthy stream although the examination time is only two. More detailed bacteriological examination will be continued from now on.

3) Ecological changes of aquatic lives neighboring the outlet of the stream and changes of environments around the equipment

Many peoples dwelling at the neighboring of the stream recognise the ecological changes in the stream from their observations. They say that a school of many small fishes are always seen to gather at the outlet of the stream which they have never seen for a long time and a clear green algae instead of grayish ones are recognised on stones in the stream. They also say that there are a plenty of children voices being in jolly spirits around the equipment. Furthermore, irrespective of persons has become to serve for cleaning the

	Number of Bacteria (per ml)		Total Colon Bacillus (per ml)		Fecal Colon Bacillus (per ml)	
Sampling Dates	1994.7	1994.8	1994.7	1994.8	1994.7	1994.8
Inlet	140,000	180,000	1,800	1,800	350	580
Outlet	36,000	12,000	480	110	17	9
Elimination Rate (%)	74	93	73	94	95	98
Culture Condition	Standard agar culture 37°C, 48hrs.		Deso culture 37°C, 19 hrs.		m-FC culture 37°C, 24 hrs.	

Table 2: *Bacteriological examination at the inlet and the outlet.*

equipment, especially around the inlet of the stream where are easier to be blocked by rubbish such as plastic goods thrown.

SUMMARY

It was the biggest worries for the author that in spite of the completion of the equipment peoples living at the neighboring of the stream might have less consciousness of keeping clean their environments and might let more dirty water flow than before because the equipment will work to treat their disposal. Fortunately, any worries did not come into existence, but the equipment is useful to recall their attentions to the environment to their minds. Peoples become aware that environment around them become worse and worse in exchange for the convenience of their lives. They become nervous because they have no idea how to improve the environment concretely by themselves. The equipment introduced here is not a treatment plant in a large scale, but can put it in place voluntarily in Japan. This is one trial how to join the activity for the protection of environment.

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POLICY AND STANDARDS

ACHIEVING SUSTAINABLE USE OF ON-SITE DOMESTIC WASTEWATER SYSTEMS

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ABSTRACT

On-site domestic wastewater systems are private facilities subject to owner/occupier management within the boundaries of the property being serviced. To achieve sustainable environmental and public health performance from on-site systems requires quality implementation processes supported by the exercise of clearly defined implementation responsibilities from all practitioners involved in such wastewater servicing. Collective oversight of system performance is essential. A range of modern treatment technologies and effluent land application methods are available to facilitate effective servicing outcomes. These must be matched to sustainable land use and water use practices in which subdivisional lot sizing procedures and water use reduction measures contribute to good environmental management of the land and water resource. The proposed 1998 introduction of a joint Australia/New Zealand Standard for "on-site domestic wastewater management" is to provide a new management framework for on-site servicing for both countries. However, successful achievement of sustainable on-site wastewater servicing, although assisted by the availability of modern technology and standardised methods of design implementation, will only eventuate when committed and informed management is exercised by all parties having responsibilities within the implementation process. This will require a coordinated approach to implementation, on-going operation, maintenance and monitoring, and regular information transfer between local authorities and all other participants in on-site wastewater servicing.

KEYWORDS

On-site wastewater; septic tanks; soakage fields; effluent management.

INTRODUCTION

On-site domestic wastewater systems are utilised for residential, institutional and commercial development in localities without community sewerage servicing. Domestic wastewater includes human wastes collected by either waterborne or non-waterborne methods together with washwaters associated with kitchen, bathroom and household laundry activities. On-site management applies to collection, treatment and land application of domestic wastewaters within the property boundaries of their place of origin.

By sustainable use of on-site domestic wastewater systems is meant the application of un-sewered wastewater servicing in a manner which achieves environmental and public health performance requirements appropriate to the cultural, economic and social objectives of the community at large, and which also takes into account the reasonably foreseeable needs of future generations. This means permanent solutions based upon the use of appropriate technology to achieve sound management of the land and water resource.

The parameters of sustainability are thus driven by the historical development of on-site wastewater servicing. In the less industrialised cultures low water use technologies for urban and rural areas drive servicing solutions. Dry-vault toilet systems with simple greywater soakaways are compatible with communal wells or rainwater supplies for low intensity domestic water use. Water closet and high consumption water use facilities in the more industrialised cultures such as in Australia and New Zealand result in on-site wastewater systems utilising treatment units coupled to a variety of effluent land application systems and matched to soil and site conditions. High volume water use requirements are supported by large storage rainwater systems, individual borewater, or centralised community water supplies.

To be sustainable, on-site wastewater systems must utilise both the water and the land resource efficiently and within their capacity to provide secure supply and assimilative services for present and future needs. This will mean effective management of on-site wastewater system implementation via sound development and planning processes, informed site evaluation, state-of-the-art design and installation practices, regular operation and maintenance procedures, and appropriate environmental monitoring controls. Whereas on-site systems have traditionally been subject to owner/occupier control of operation and maintenance, sustainability objectives make it imperative that collective oversight of system performance be instituted to support owner/occupiers in achieving long-term reliability of wastewater servicing.

PERFORMANCE REQUIREMENTS

The most important performance requirement for on-site wastewater management is protection of public health. This requires control of human wastes in a manner that prevents the spread of infectious agents (pathogenic bacteria and viruses) via land or water. Furthermore, accumulation of effluent residuals such as nitrates in potential sources of water supply must be avoided.

The second most important performance requirement is maintaining and enhancing environmental quality. This means the assimilative capacity of soil, water and vegetation resources must not be overwhelmed by the discharge of organic matter, nutrient and other salts (including nitrates, phosphates and sodium) nor of liquid volume.

Community amenity must also be maintained and enhanced by ensuring on-site systems are technically robust and of long life, have economy in operation and refurbishment, are nuisance free in terms of odour and/or noise, and provide a permanent solution for wastewater servicing.

The final performance requirement relates to achieving efficiency in resource use including land, water and energy. This requires minimising demand on land and water resources and maximising, where practicable, the utilisation of wastewater solid and liquid nutrient materials for beneficial environmental uses.

The achievement of these performance objectives cannot be attained by the application of traditional approaches to on-site wastewater system implementation and regulatory control. These traditional approaches have been characterised by inadequate land use and development planning, poor soil and site assessment practices, flawed design loading rate choices, inadequate supervision of construction, short cuts in installation practices, negligible operation and maintenance attention, and variable application of regulatory oversight and approval processes. This has resulted in significant "failure" rates in some localities.

Sustainability performance requirements will only be achieved when:-

- quality implementation processes for planning, investigation, design, installation, operation, maintenance, and monitoring are set in place
- clearly defined implementation responsibilities for all practitioners and participants in on-site wastewater servicing are set out and adhered to.

Education, training and comprehensive information services will be the foundation upon which such implementation processes and responsibilities achieve their full potential in meeting performance requirements.

ON-SITE TREATMENT TECHNOLOGIES

Dry-vault systems for non-waterborne human waste collection, consolidation, and stabilisation have evolved from the traditional pit privy through to modern composting toilets. A variety of alternative dry-vaults have been developed to meet appropriate technology objectives for wastewater servicing. The United Nations Environment Programme (UNEP) water supply and sanitation decade of the 1980's led to the development of the VIP (ventilated-improved-pit) toilet for use in less developed countries as an economical and user-friendly dry-vault. The VIV (ventilated-improved-vault) toilet is an adaptation of the VIP which relies on windshear venting via a 300 mm diameter vent pipe for cooler wetter climate use. The VIP 150 mm vent pipe system is applicable in warmer dryer climate conditions where thermal venting is reliable. Hence VIV toilet systems have been adopted in more developed countries where pumpout servicing and off-site solids treatment, stabilisation, and disposal facilities can be provided.

Compost toilets provide a controlled environment to manage human waste via combined aerobic/anaerobic processes with a stabilised product output to be removed periodically. However, the regular daily/weekly commitment to managing the composting operation means that only the most dedicated user finds such systems an acceptable alternative to conventional waterborne systems.

Waterborne flush toilet (water closet) systems are a product of western cultures which have led to provision of community sewerage systems and centralised wastewater treatment and disposal. The on-site version of waterborne servicing traditionally developed around the septic tank, a simple chamber in which scum and solids from domestic activities accumulate. The settled outflow is then passed to a subsoil soakage system, traditionally shallow, gravel filled trenches capped with topsoil.

Conventional septic tanks are giving way to high performance tank units incorporating effluent outlet filters to control solids. The use of modern plastic mesh screens in tube formation, or of stacked disk settling units, can provide a significant improvement in solids control. This has two benefits. First, it protects the subsoil treatment and soakage system from excess solids carryover, thus increasing system life and enhancing sustainability performance. Second, outlet solids control systems enhance settling performance, enabling better consolidation and stabilisation of scum and sludge thus increasing storage capacity and allowing longer periods between solids pumpout. Six to ten year pumpout desludging of the larger capacity high performance septic tanks may be achievable compared to three to five year pumpout frequencies advocated for conventional tanks.

Alternative domestic wastewater treatment systems based upon scaled down versions of community treatment plant biological processes include AWTS (aerated wastewater treatment systems) and sand filter units. Substantial improvement in treatment level compared to septic tanks is achieved by such systems. Organic matter removal is the principal improvement, with sand filter systems producing a higher quality and more reliable effluent that AWTS. Although the higher effluent quality is a direct contribution to improved sustainability of on-site wastewater systems, this comes at a price in terms of capital, operating, and maintenance costs. However, the priority for environmental sustainability may in some development situations be of higher importance than economic factors. Additional technical features can be provided to achieve disinfection for bacterial control, and nutrient removal for public health and environmental control. Variations on AWTS units involve activated sludge and biofilter technologies; variations on sand filter units involve intermittent pulse dosing and recirculating technologies.

ON-SITE LAND APPLICATION METHODS

Traditional septic tank and soakage trench systems have often been utilised in urban fringe subdivisional development as a temporary measure until full community sewerage servicing is provided. Public authority commitment to improving on-site system technology and operation and maintenance was negligible during this era. During the 1960's in the USA, where a higher proportion of unsewered development occurred compared to Australia and New Zealand, a considerable research effort was made into alternative land application systems ("disposal" systems) for septic tank effluent. This produced a range of new systems for improved in-soil treatment compared to traditional trenches and beds. Evapo-transpiration trenches and beds, sand mound systems built over existing topsoil layers, sub-surface irrigation via pump dose loaded mini-trenches, and surface spray irrigation systems coupled to AWTS or sand filter treatment, all entered the technology choice available to designers. Each system has its own range of site, soil and environmental conditions for which it is best suited, and the challenge for designers in achieving implementation of a sustainable servicing solution in each case is to select the most appropriate system, and then apply loading rate values to achieve optimum design sizing. Because implementation of land application systems for on-site wastewater effluents is more an art than a science, achieving sustainable solutions is dependent less on selection of design sizing values against scientifically determined soil and site characteristics, and more upon building in factors of safety for system operation. These are required to allow for wide loading fluctuations due to variation in occupier numbers and water use habits, deficiencies in installation practice, misapplication of design loading rate choice against site and soil information, and neglect in operation and maintenance due to owner/occupier inattention. The ultimate backup to ensure sustainability of land application in such circumstances is to provide 100% reserve area on-site to facilitate complete system replacement if or when the original system "fails". When faced with the cost of replacement, owner/occupiers become more conscious of the importance of on-going protection of their investment by taking a pro-active interest in operation and maintenance. Reserve area requirements could be reduced where mechanisms to ensure operational security are set in place, such as community wide operation and maintenance schemes.

SUSTAINABLE LAND USE

Traditional on-site wastewater practices in Australia and New Zealand for rural-residential subdivision have centred around lot sizes of 1/4 acre (1000 m^2) reduced to 1/5 acre (800 m^2) or smaller in sandy soil conditions such as beach resort areas. Developers seek always to maximise lot density while also maximising effluent loading into subsoil and groundwater. Where groundwater utilisation for household water supply is a constraint on achieving maximum lot density due to potential contamination from bacteria and nutrients, either lot sizes are increased to dilute the effluent loading, or a community water supply scheme is provided to avoid reliance on individual groundwater sources.

To make efficient use of the land resource means maximising lot density; to achieve sustainable use of the assimilative capacity of the environment means minimising effluent loading density. These incompatible objectives can be reconciled in one of three ways.

First, use of conventional septic tank and soakage field systems would only be permitted on large lot sizes (2500 m^2 or larger has been adopted in some localities in New Zealand). Second, smaller lot sizes would be permitted where improved on-site treatment and land application technology is used. This improved technology can extend to disinfection for bacterial control, and nutrient stripping for both public health (eg nitrates in drinking water) and environmental control.

Third, subdivisional layout can be re-configured into "eco-village" format by which individual dwellings are clustered onto smaller (500 m^2 to 800 m^2) lot sizes within the least favourable soil areas while the remaining better soils are retained for communal effluent disposal. High performance septic tanks would be provided on each lot with small diameter sewer collection of effluent to a communal secondary treatment plant (eg recirculating sandfilter) with surface trickle or subsurface dripper irrigation into landscaped "green-area" communal land application. A community management entity such as a body corporate is required to implement and maintain this form of communal on-site system to ensure the improved environmental performance of this approach over individual on-lot effluent management is sustained. Such an "eco-village" development model lends itself to a new form of rural urbanisation in which potential exists for integrating residential, horticultural, orcharding, livestock production, light industrial, commercial and recreational land uses whereby environmentally sustainable wastewater servicing is an integral component of such development.

A recent study by Gardner et al (1997) into environmentally sustainable on-site effluent treatment systems for Australia addresses the lot density question in some detail. Nutrient loadings in terms of public health (nitrates) and environmental effects (nitrates and phosphates) are determinants of lot size. The study points out that present technology practices and design rules have resulted in significant "failure" rates of both conventional septic tank trench systems and AWTS plus effluent spray irrigation. To accommodate potential environmental effects from such practices lot density must be reduced. Increase in lot sizes to achieve environmental sustainability (where groundwater contamination is not an issue, i.e. community water supply is available) dictates 4,000 m^2 to 10,000 m^2 minimums for septic tank and soil absorption systems, and 3000 m^2 to 5000 m^2 minimums for AWTS and irrigation systems.

Such requirements for environmental sustainability of on-site systems are clearly very consumptive of the land resource. To reduce lot sizes and achieve efficient land use while maintaining long term environmental quality will require a change in on-site implementation practices. This will include use of improved treatment technology and land application methods as well as varied development configurations such as "eco-village" concepts. However, such measures will only be successful to the extent by which they are supported by effective management oversight.

SUSTAINABLE WATER USE

There is heightened public awareness of the need to manage water resources more effectively, and this is reflected in the availability of products and fixtures for reducing household water use. These include reduced flush water closets, low flow shower heads, aerator faucets, front load washing machines, and pressure/flow control inserts on faucets. As well as reducing water consumption, such savings measures reduce household energy use (via lower hot water requirements) and result in lower wastewater outputs for on-site treatment and disposal via land application. Further conservation of the water resource can be achieved by greywater reuse. Greywater from bathroom and laundry use can be stored for garden and landscape watering in dryer climates. Alternatively, greywater can be treated and recycled for water closet flushing thus resulting in an immediate one-third reduction in household wastewater output.

The implications of maximising wastewater reduction on the sustainability of on-site systems is significant. Land application areas can be reduced in size commensurate with lower flow volumes, although it must be recognised that overall effluent residuals will not be reduced in proportion, but only to the extent that lower flow volumes contribute to improved treatment performance. Land application systems will thus have to accommodate lower flow volumes but potentially stronger effluents, requiring more effective loading arrangements such as dosing and resting. Full greywater recycle coupled with maximum use of water reduction fixtures can achieve 50 to 60% wastewater reduction in a domestic household situation. The Auckland Regional Council in New Zealand allows a concession on on-site land application design sizing to reflect such flow reduction provided that high performance septic tanks or better pre-treatment is used, and that a 100% reserve area is set aside for the land application system extension, if needed in the future. However, this still achieves significant reduction in landuse and water use as a contribution to environmental sustainability. Management and regulatory monitoring requirements have to be stepped up to ensure effective oversight of such measures.

MANAGEMENT THE KEY TO SUSTAINABILITY

The proposed joint Australia/New Zealand Standard "On-site Domestic Wastewater Management" which is due out in 1998 sets the scene for a new management framework. The underlying purpose of the joint Standard is to identify and give guidance on the implementation processes and management responsibilities within those processes that are necessary to achieve sustainable environmental and public health performance from on-site domestic wastewater systems. Performance requirements are set that aim to ensure on-site wastewater servicing practice is undertaken in a manner which protects and enhances community health, is ecologically sustainable, and achieves environmental management objectives for air, land and water resources.

Implementation of on-site wastewater servicing is to be undertaken in accordance with appropriate standards for site evaluation, design, installation and construction, operation and maintenance, and education and training. Successful implementation is also dependent upon all persons involved in on-site systems carrying out their responsibilities in a diligent and informed manner. Such persons include planners, surveyors and land developers, site evaluators/soil assessors, designers, installers and contractors, equipment manufacturers and suppliers, pumpout/de-sludging contractors, regulatory authorities, homeowners and occupiers, estate agents and property transfer lawyers.

The "management" Standard also provides technical guidelines involving conservative design rules for sizing septic tank systems and land application methods (including trenches, beds, evapo-transpiration-absorption systems, disposal mounds and irrigation systems).

New product Standards for septic tanks, greywater tanks, compost toilets and aerated wastewater treatment systems make up the suite of documents governing on-site system implementation and management for the future.

However, success in terms of achieving sustainable use of on-site systems does not lie with the content of these new Standards, but with the diligence by which all persons regulating, designing, installing, servicing and using on-site systems undertake their responsibilities. The key to ensuring responsible oversight of on-site systems is education, training and information sharing. The current level of unsatisfactory environmental performance of on-site systems is a reflection on the lack of co-ordinated information transfer. The new Standard advocates specific training groups for the range of players involved in on-site wastewater servicing. Each group will participate in three levels of training which include study of the basic principles of on-site systems and their management, application training specific to the group members role in the implementation process, and regular refresher training on new technology practices and procedures.

Regulatory authorities are seen as having a most significant role in monitoring and co-ordinating training and information sharing activities. The holding of regular forums which on a district or regional basis bring together all players involved in on-site wastewater servicing to participate in information sharing, problem solving and cross-disciplinary contact will enable better understanding and co-operative liaison to be developed between regulators and practitioners.

Effective operation, maintenance and monitoring procedures are the other essential element to achieving sustainable performance outcomes for on-site wastewater servicing. The initiative to see that this happens lies firmly with regulatory authorities. Three administrative approaches are suggested as applicable depending upon local and regional circumstances. First is a local authority administered scheme for septic tank pumpout and system inspection to a regular schedule. This approach is particularly suitable for permanently occupied localities. Second is a local authority administered inspection scheme, particularly suited to intermittently occupied dwellings as in holiday resort areas. Third is the body corporate approach where the community sets up its own operation, maintenance and inspection protocols to be undertaken by employee or contract arrangements for achieving treatment tank de-sludging, equipment upkeep, system upgrading and environmental monitoring. Commercial services which will design, build and operate on-site systems for subdivisions and/or "eco-village" developments already operate in New Zealand.

In the absence of such operation and maintenance measures at the local level, self-monitoring by owner/occupier is essential. This can be fostered by local authority by-law requirements which provide for regular "warrant-of-fitness" checks of on-site systems. Hence owner/occupiers would be required to engage at their own cost the services of a qualified and experienced person to undertaken an operation and maintenance check at regular intervals (say 12 months, or 2 years). The regulatory authority would maintain the database records for its area, and issue reminders for such monitoring checks to be undertaken. One District Council in New Zealand is well advanced with planning to institute the "warrant-of-fitness" approach to on-site system management.

CONCLUSION

Achieving sustainable use of on-site domestic wastewater systems will mean a significant change from past practices which have perceived on-site systems as but temporary solutions until conventional wastewater servicing is implemented. There is a current demand for decentralised rural/residential development in many localities in Australia and New Zealand which the community expects to be serviced in an environmentally sustainable manner. Land application of wastewater effluents treated on-site can achieve such objectives provided that appropriate performance requirements are set in place, and that technical and administrative implementation measures evolve to ensure reliable and consistent achievement of those requirements.

New treatment and land application technologies are available to support such endeavours and the proposed Australia/New Zealand Standard sets in place the management framework for advancing on-site system sustainability. However, successful implementation will only be assured by committed and informed participation by all parties having responsibility for each element of on-site wastewater servicing. This is the future challenge for all on-site wastewater system practitioners.

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THE PURSUIT OF EVER MORE STRINGENT STANDARDS FOR EFFLUENT QUALITY

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INTRODUCTION

For several years ACTEW Corporation has been investigating alternative methods of delivering its traditional water, sewerage and electricity services. One alternative being trialed is the on-site recycling of domestic wastewater. The focus has been on beneficial re-use of this treated effluent to defer the development of new potable water sources, and to minimise the cost of developing and servicing centralised sewerage infrastructure. Traditional sewerage development in Canberra involves disposal of ever growing quantities of wastewater to the Murray-Darling system. There are large economic and environmental costs associated with this method of disposal. This implies that serious attention to recycling wastewater is an imperative, not some fashionable luxury.

ACTEW has recognised that a large proportion of our total water consumption is being used for irrigation and flushing toilets. These needs could easily be met by using recycled water. Indeed, there is an environmental advantage of beneficially using the contained nutrients on land where they can displace the need for manufactured fertilisers. There is then the added advantage of keeping these same nutrients out of our streams and rivers where they are causing harm.

One method of recycling water is to add another pipe network system to the existing water, sewer and stormwater hydraulic networks and bring the treated effluent from the centralised sewerage plant back to the community. However, we realised that we were already spending about eighty per cent of the total servicing cost on moving water from one place to another. The possibility of treating the water on-site, doing away with two hydraulic systems, and even reducing the size of the water supply and stormwater network, would appear to be very attractive on all counts.

ACTEW's EFFLUENT RE-USE TRIAL

Six Canberra households were disconnected from the sewerage system over the summer of 1994/95. All wastewater was diverted to individual domestic scaled treatment plants. At each site, the treated effluent was disinfected using chlorine and pumped to a fourteen kilolitre holding tank. From there it was used for irrigation purposes and to flush internal toilets. The big advantage with this system centres around the storage tank. A waste product becomes a valuable resource. By having storage, the effluent can be used *when* and *where* it is needed. The treated effluent is used to water the whole garden, rather than simply disposing of it in a corner, and then using potable water to irrigate the rest. The owner also has control to choose when irrigation will take place. The most appropriate times are when evaporation is low, when it is not raining, and at night when people are not around. By using a programmable controller, irrigation can be automatic.

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The effluent is also available through specially marked and signed garden taps for general watering purposes. Householders are advised to minimise direct contact with the water and children should not play in the water. They were also advised, as a precautionary measure, that they should close the toilet lid when flushing the toilet.

The trial has now been going for almost three years and all of the householders have maintained a keen interest in the trial. All want to remain on the program even though at times they have [1]been inconvenienced by occasional plant breakdowns. In a few cases the effluent quality has not always met the very high standard of less than 10 colony forming units (cfu) per 100 ml for thermotolerant coliforms required by the ACT Health Department.[1] ACTEW Corporation, as part of its commitment to protecting public health, engaged the ACT Health Department to closely monitor the systems as a back up to the routine monitoring which we have undertaken. ACTEW has also commissioned CSIRO to investigate the extent of groundwater leaching of salts and nutrients. It will also investigate the value of the wastewater particularly in terms of closing the water and nutrient cycles. These findings are further being compared with traditional methods of watering and fertilising gardens.

As a result of the trial, all volunteers are now much more aware of the water and nutrient cycles that are occurring on their blocks and are much more mindful of what and how much they flush down their drains. Many of the traditional chemicals once stored under the laundry sink, have left the scene. Some home owners have even commented that they have rediscovered traditional cleaning agents such as sugar soap and vinegar.

WASTEWATER REUSE REGULATION

The current Australian approach to dealing with wastewater reuse has been through the use of non-mandatory guidelines. State and territory authorities then impose standards through a licensing arrangement based on local conditions and treatment objectives. However, in the absence of any other research or experience, and possibly current industry practice, regulatory authorities appear to be adopting these guidelines as defacto standards. This is despite the comment by the authors of these guidelines that "the guidelines have not been developed for regulatory purposes and the values given should not be construed as standards".[2]

THE QUESTION OF STRICTER STANDARDS

Australia seems to be heading down the conservative path of increasingly stringent microbiological criteria or "zero risk" as a method of reducing the potential for disease from reusing wastewater. *Guidelines for the Use of Recycled Water in NSW*[3] was published in 1978 by the NSW State Pollution Control Commission for use of municipal wastewater. It outlined three levels of disinfection, the most stringent being for irrigation of landscaped areas with public access and irrigation of crops for human consumption. This level of protection required the geometric mean of at least five samples to be less than 300 cfu faecal coliforms per 100mL with an upper limit of 2000 cfu per 100mL. For the purposes of this discussion faecal coliforms and thermotolerant coliforms are considered equivalent measures. With public exclusion during spraying these levels were relaxed to a geometric mean of less than 750 cfu per 100mL and an upper limit of 5000 cfu per 100mL.

In 1987, the *NHMRC/AWRC Reclaimed Water Guidelines*[4] were published. Intended primarily for municipal wastewater reuse, the guidelines required secondary treatment and disinfection so that the faecal coliform count did not exceed "a median value of 1000 cfu per 100 ml for five samples collected at not less than thirty minute intervals with four out of five samples containing less than 4000 cfu per 100 ml". If the effluent were to be used for spraying salad crops (ie crops eaten without peeling or cooking) then the corresponding limits were set to 10 and 20 cfu per 100 ml respectively.

Since that time a number of new standards and guidelines have emerged with increasingly stringent requirements. As part of the development of a National Water Quality Management

Strategy, draft *Guidelines for Sewerage Systems - Use of Reclaimed Water* was produced in April 1996.[5] These guidelines superseded the NHMRC (1987) guidelines. The guidelines recommend acceptable levels of treatment, chlorine residuals, turbidity, biochemical oxygen demand (BOD), suspended solids and thermotolerant coliform counts for various uses. An upper limit of 10 cfu per 100 ml has been established for urban non-potable residential use (eg gardening), while the upper limit for primary contact (eg swimming) is 150 cfu per 100 ml. Low contact activities such as horticulture have a limit of 1000 cfu per 100mL. The *ACT Wastewater Reuse Guidelines*[6] (1996) have followed this trend by placing the limit for thermotolerant coliforms for municipal irrigation with uncontrolled public access, and residential use, at a median value less than 10 cfu per 100mL with four out of five samples less than 40 cfu per 100mL.

Where, and on what basis, are these upper limits set? Why can we swim in water fifteen times more polluted than we can water our garden with? Is there any epidemiological basis for setting these Australian guidelines or are they simply based on what appears to be reasonable in the mainstream? A 1989 World Health Organisation publication *Health Guidelines for the use of wastewater in agriculture and aquaculture* stated: "based on current epidemiological evidence, a bacterial guideline of a geometric mean of 1000 faecal coliforms per 100 ml for unrestricted irrigation of all crops is recommended." There is no question that untreated wastewater is a health hazard so disinfection is necessary. However the infectious risk hazards need quantifying.

The 1992 publication *Australian Water Quality Guidelines for Fresh and Marine Waters*[7] suggests that farmers can irrigate crops with river water containing up to 1000 cfu per 100mL. For raw food crops irrigated with wastewater, the wastewater must contain less than 10 cfu per 100mL. A survey of some river water quality data[8] along the Murray River suggests *E.coli* levels of between 0 and 100 are quite common in dry weather and even higher in wet weather. Does this mean that river water should receive further disinfection to 10 cfu per 100mL when irrigating raw food crops?

The *NSW Guidelines for Urban and Residential Use of Reclaimed Water*[9] (1993) were written to apply to reticulated reclaimed water systems under ownership of a central supply authority. These reduce microbiological levels further to less than 2.5 faecal coliforms in 100mL, and also specify coliform, virus and parasite requirements. Despite the high level of treatment and resultant wastewater quality, irrigation of food crops was still not recommended. Does this indicate Australia is heading in the same direction as the California Reuse Guidelines which require a median value of less than 2.2cfu per 100mL for unrestricted urban reuse?[10]

Adequate disinfection is definitely needed to maintain good public health, but what is the evidence that supports the increasingly stringent effluent standards? Is there any evidence that not adhering to the NHMRC (1987) guidelines was causing a serious community health problem? Has any epidemiological study demonstrated that communities using municipal wastewater on their parks and golf courses before 1987, or those using aerated septics (which we know often fail even current standards), suffered from a lower standard of health than others connected to a conventional sewerage system?

Guidelines and standards specific for reuse of wastewater from on-site systems have only come to fruition recently. These appear to be even more stringent than municipal wastewater. The draft *Environmental and Health Protection Guidelines: On-site Wastewater Management Systems for Domestic Households* [11] suggests that the expected design quality of wastewater from an Aerated Wastewater Treatment System (AWTS) will achieve less than 30 cfu per 100mL for faecal coliforms. Yet, AS1547-1994 *Disposal systems for effluent from domestic premises*[12] suggests "for information only" that "samples taken on each day shall have a thermotolerant coliform count not exceeding a median value of 10 cfu per 100mL with four out of five containing less than 20 cfu per 100mL."

Are regulators trying to discourage the use of on-site systems, or encourage the use of more chlorine? Surely, treated wastewater emanating from a single household is inherently safer for that family (and even their neighbours on rare occasions) to come in contact with than treated wastewater collected from the whole community. Is it hoped that some trickle down effect will work here, so that if the standard is set high enough, then the quality of the worst plants will also improve to something that is reasonable? If this is true, what is considered reasonable and is this the best way of achieving it across the board?

RISK ASSESSMENT

As previously mentioned, the ACTEW recycling trial involved the use of recycled water from the domestic treatment plant for toilet flushing. Concerns were expressed about the health implications of using recycled water in the confined space of a toilet room as aerosols could easily be inhaled or come in contact with people flushing the toilet. Reflecting on this concern, it seemed likely that the quality of the water in the toilet bowl had little to do with the quality of the water supply, but more to do with the contents of the bowl! On two separate occasions samples of *ostensibly* clean water were collected from the bowls of flushed toilets in our office and analysed. The lowest *E. coli* count from the fifteen composite samples analysed was 3 cfu per 100 ml while the highest was 780,000 cfu per 100 ml. The median count for the contents of the male toilets was 2500 cfu per 100 ml while that for the female toilets was only 540 cfu per 100 ml. This lead to idle speculation that the standard of hygiene shown by the women were perhaps greater than for the men! However, a more likely explanation is that the difference can be attributed a higher proportion of men in the office, and a reduced amount of flushing of male toilets because of the use of urinals.

On the basis of the current standards or guidelines, banning the use of flushed toilets should be seriously considered as they would appear to constitute a major health risk. Perhaps an interlock should be mandated which prevents flushing unless the lid is closed? Given that most people don't close the lid when flushing, why hasn't this apparent health problem been observed before this? Many public toilets do not even have a lid and these are the ones most likely to be grossly contaminated. One may wonder why are we required to treat wastewater for flushing toilets to 10 cfu per 100mL with test results like these. Are we spending too much time and money unnecessarily reducing risk in one area when it would be better spent in other areas?

THE COMMUNITY COST OF MEETING STANDARDS

Increasingly, the community is questioning how Governments are raising and spending their money in order to bring about change. Generally, the money they spend is raised through taxation and governments are often made and lost on the basis of their taxation policies. Governments can also bring about change by introducing regulations. Meeting the requirements of new regulations costs the community dearly but the cost does not come from taxation. The cost is met mainly from the increased cost of the products they purchase. To date there has been little public scrutiny of these costs, but this is set to change. The community has started to question the costs and benefits of an ever growing regulatory impost. It was estimated by *The Economist* (July 27, 1996) that in 1995, the average American household paid out an additional $7000 to meet costs imposed by regulations compared to an average tax bill of only $6000.

Has any analysis been done to estimate the environmental, health and dollar impacts of meeting the higher standards that are proposed in the most recent guidelines? It seems likely that, in the future, regulators will need to be able to demonstrate how and why the various regulations are determined and to engage the community to a much greater extent in the setting of standards.

By way of example, the World Health Organisation has recently taken the USEPA to task over its overly zealous approach to the treatment of wastewater.[13] For developing countries this has meant expenditure of scarce dollars on extended water treatment for irrigation practices that present little health risk. These practices are mandated at the expense of protecting and treating critical potable water supplies.

Closer to home in the west, we are advocating sustainability, but at the same time we *may be* discouraging optimal use of water resources by placing unreasonable cost impositions on water treatment. A good example is the irrigation of country town golf courses, an established practice of nearly a quarter of a century. There have been no claims of adverse health effects but the suggested turbidity requirement in *Guidelines for Sewerage Systems - Use of Reclaimed Water*

(1996) points to the need for expensive filtration. Is this diverting scarce dollars away from other more pressing health and environmental problems in need of a solution?

THE QUESTION OF HEALTH

Does compliance with ever more stringent effluent guidelines or standards really improve public health? Maybe, maybe not. It depends how 'health' is measured.

The 1948 WHO definition was :-

> *"Health is a state of complete physical, mental and social well-being, not merely the absence of disease or infirmity"*

On the basis of this definition, there would appear to be very little evidence that a requirement to constantly increase the quality of effluent improves our health. In fact, in order to meet ever more stringent standards, requires the use of either more powerful or greater quantities of biocides. These chemicals *may* be having a serious adverse effect on our health as well as on the health of the environment. The ecological health of our environment is vital to our survival. Under the NHMRC (1987) guidelines, ponding of treated effluent was encouraged. Ponding alone would not be sufficient to meet current recommendations.

ACTEW has been trialing the use of ultraviolet radiation (UV) to disinfect treated effluent at a domestic scale. UV disinfection is viewed as being more environmentally benign than chlorine. Whilst it has been possible to meet the present guidelines during winter when temperatures are low, the absence of a disinfectant residual has meant that it has not been practical, or always possible to meet these limits in summer due to regrowth. It may one day be shown that the environmental and health problems caused by the increasing use of chlorine may be far more damaging than the benefits attributed to a more sterile effluent.

Does it even follow that a more sterile environment leads to a more healthy community? Many diseases now cause greater community morbidity because children now are often not exposed to them until they are older when they are less able to cope with the condition. Our immune system (on which we are all totally reliant if we want to have any quality of life) needs regular stimulation if it is to remain in good working condition. This does *not* mean being overwhelmed by pathogens, but where is the evidence that the previous NHMRC (1987) guideline value of 1000 cfu per 100 ml was causing a problem? In fact, many American studies as discussed by Shuval[14] have shown that sewerage workers and residents living in close proximity to activated sludge plants (ie those who live and work in conditions much worse than anything suggested here), showed no significant adverse health affects that could be attributed to their proximity to aerosol borne pathogens from these plants.

The following paragraphs are quoted from the World Health Organisation's publication, *Health Guidelines for the Use of Wastewater in Agriculture and Aquaculture* (1989):

> *"The consensus view of the epidemiologists and public health experts who have reviewed these data is that the actual risk associated with irrigation with treated wastewater is much lower than previously estimated and that the early microbial standards and guidelines for effluent to be used for unrestricted irrigation of vegetables and salad crops normally consumed uncooked were unjustifiably restrictive, particularly in respect of bacterial pathogens. The epidemiological evidence is summarised in this report; for more detailed information, readers are referred to the original reports by Shuval et al.[15] and Blum and Feachem[16] and to the Engelberg[17] report."*

The situations cited in these studies would seem to be vastly poorer than the conditions presented by an on-site domestic treatment unit. Here the occupants are, at worst, exposed to pathogens that have been passed on by family or friends or possibly neighbours. It would then seem likely that antibodies already developed against these pathogens from normal social interactions, would work to counter these same pathogens acquired from low level contact with the treated and disinfected domestic wastewater.

One might also argue that there is a much greater public health risk presented by wastewater centrally collected and returned to the community for reuse purposes. It is recognised that a failure of the central system could potentially distribute pathogens from a local source to the *whole* community via the reticulation system. In contrast, a partial failure of an on-site system can, at worst, only spread pathogens (if they are present) to those in the immediate vicinity.

These risks need to be contrasted with the very high, but accepted, community risks from poisoning, electrocution and from driving your car. However, we all expose ourselves to these risks because we feel that the benefits outweigh the disadvantages. It would indeed be unfortunate if local recycling with all of its potential benefits, both in economic and environmental terms, could not be practiced because of arbitrarily stringent guidelines and standards that appear to have little relevance to the overall well-being of the community.

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NEIGHBOURHOOD WASTEWATER TREATMENT PLANTS ARE CHEAPER: FACT OR FICTION?

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ABSTRACT

This paper examines two alternative approaches of wastewater treatment systems, large centralised systems and smaller localised neighbourhood treatment systems, with special emphasis on the situation facing the Perth metropolitan area. It looks at the costs associated with the land acquisition, capital works, operating and maintenance expenditure, the provision of buffer areas and methods of effluent disposal. It also discusses the problems of acquiring land and gaining approvals for treatment and reuse of effluent and biosolids as well as health, social and environmental implications of the two types of systems. In the comparison a centralised treatment plant is assumed to be one with an equivalent population (ep) of 400 000 persons and a neighbourhood treatment plant assumed to have an equivalent population of 10 000 persons. On this basis, Perth would have, on current population, three central treatment plants or 120 neighbourhood plants.

INTRODUCTION

Traditionally large cities have constructed major centralised wastewater treatment plants, with disposal of treated effluent to receiving waters such as streams, rivers, lakes and oceans. This approach had been adopted in the belief that such facilities were easier and cheaper to operate than small neighbourhood treatment plants which disposed of effluent locally.

Recently this view has been challenged, particularly with regard to the cost of providing main sewers, operating costs and methods of disposal. Small neighbourhood plants with reuse of effluent are perceived as being viable alternatives.

The metropolitan region of Perth covers a large area and the majority of urban development is located on the relatively flat Swan Coastal Plain. Groundwater is close to the surface and gravity sewers are difficult to lay in the very wet soil conditions frequently encountered. Due to the ever increasing suburban sprawl, sewer mains are very long. With Perth's high ambient temperatures and extensive use of pressure mains, the wastewater tends to be very septic and quite odorous.

For these reasons, in the 1960's, a decision was made to try to treat the wastewater in Perth as close to the source as possible. Many permanent neighbourhood wastewater treatment plants were constructed. Locations where these types of systems were installed include Kelmscott, Westfield, Canning Vale, Kwinana, Two Rocks, Yanchep, Point Peron, Port Kennedy and Swanbourne. Many of these have since been decommissioned and all will be phased out over the next 5 - 10 years. It has been decided to concentrate on constructing centralised coastal wastewater treatment plants, producing a high quality effluent suitable for both reuse and return to the ocean. Disposal of reclaimed water to the land will be maximised and a reuse policy will be developed over the next few years.

Options include provision of dual water supplies, agroforestry, industrial reuse, irrigation of ovals and use in horticulture.

The main reasons for the swing around in philosophy have been;

- good septicity control in sewers using oxygen injection and chlorine injection,
- problems in acquiring wastewater treatment and effluent disposal sites,
- effluent disposal problems, especially in winter,
- cost of operation of localised neighbourhood wastewater treatment plants, and
- restrictions on effluent disposal due to the Underground Water Pollution Control Areas.

Despite the decision to continue to use large centralised sewerage systems, the public have recommended that the Water Corporation regularly review this policy. It is also recognised that neighbourhood treatment plants provide the opportunity to trial new technologies and are often the preferred option for sewerage systems in relatively isolated communities on the fringes of the Perth metropolitan area.

1.0 CAPITAL COSTS

The capital costs of wastewater treatment plants, both large centralised plants and small neighbourhood plants, include the cost of investigations, approvals, design, construction of the collection system pipes and pumping stations as well as the treatment plant and disposal systems, and any ancillary works such as roads, fences, workshops, offices etc. Capital costs also usually include the cost of land acquisition including the buffer zone. In this comparison, land acquisition will be discussed separately. The costs of the treatment and disposal will also be looked at separately from the costs of the collection system.

1.1 The Collection System

The primary difference between the collection system for a large centralised treatment plant to that of a smaller neighbourhood treatment plant, is the need for main sewers. For large centralised systems there is a main sewer system which collects wastewater from a number of reticulation systems and conveys the wastewater to a large central treatment and reuse/disposal facility. The main sewer system is developed as the catchment grows and its capacity can be increased to accommodate larger flows. For smaller neighbourhood systems, the reticulation system conveys the wastewater straight to the treatment facility.

The main sewers for the Subiaco Wastewater Treatment Plant catchment have a replacement cost of \$150 M with an average life of 50 years. Taking into account a depreciation value of \$2.5 M/year, this gives a unit cost of 14 cents/m^3. A neighbourhood system would not have this unit cost built in to the overall capital costs.

In a comparison made between using a piped reticulation system, conveying the wastewater to a central wastewater treatment plant by a combination of gravity and pumps along a main sewer and a system whereby the wastewater would be collected via pipes and conveyed to a neighbourhood treatment plant, for a development comprising of 120 dwellings on a 8.8 ha area, it was found that the actual reticulation costs component of the former were slightly cheaper due to a lower excavation depth required. The capital costs were \$ 143 000 compared to \$150 000. (UWRAA 1997).

1.2 Treatment and Disposal

The treatment and disposal costs for a neighbourhood treatment plant serving an equivalent population of 10 000 people would obviously vary depending on the treatment method chosen and the options for reuse and disposal. However, given the land area requirements for more passive systems such as oxidation ponds and Ecomax ™ type systems, these would generally not be a viable option for the metropolitan area. To treat the wastewater to an adequate quality, matching that of the large secondary treatment plants such as Beenyup and Subiaco, treatment technology would probably have to be similar to the more advanced wastewater treatment plants installed in some of the State's south west such as at Pemberton, Denmark and Australind. These plants consist of an Intermittently Aerated Extended Aeration process followed by some method of disinfection.

The capital cost for these types of treatment plants, for the size being examined here, would be estimated to be approximately $2.5 M. Given that for a population the size of Perth would require 120 of these neighbourhood systems, the overall capital cost would be $300 M.

A large secondary wastewater treatment plant serving an equivalent population of 400 000 people, once again depending on the technology employed, could be estimated to be in the vicinity of $100 M for capital costs. Thus for three of these installations, an overall capital cost of $300 M would be realistic. Thus the capital costs for treatment and disposal for the two options appear to be quite similar.

1.3 Land Acquisition

Each neighbourhood treatment plant would require an area of at least 2 hectares for the assets alone. This equates to a total area of 240 ha for the 120 sites required. A centralised treatment plant would demand an area of about 20 ha for the assets. Thus 60 ha would need to be purchased if utilising three large centralised systems. Obviously this would have a huge bearing on the capital cost of employing neighbourhood treatment plants over the more conventional system of large centralised treatment plants. In addition, these land areas do not include the substantial land area requirements for buffer zones.

Desired buffer distances for neighbourhood treatment plants of 10 000 ep is a minimum of 500 m from the inner plant boundary. Thus the total area requirement for buffers around all of the 120 neighbourhood plants would be 9 400 ha and this is as a minimum. A minimum buffer distance for a large centralised treatment plant is 1 km. Hence the total land area required for the three plants would be approximately 940 ha. Therefore the cost of purchasing land for buffer areas around the neighbourhood plants would be likely to be at least 10 times higher than that of purchasing land for buffer areas around the three large treatment plants.

Acquiring land for wastewater treatment and disposal has proven in the past to be a difficult and lengthy process. The purchase of two future sites in Perth, at East Rockingham and Alkimos have yet to be finalised after a decade of negotiations. Even obtaining very small sites at Port Kennedy and Mundaring took over five years for the approvals process. Approvals for treatment and reuse of effluent and biosolids also take time. Extra resources would be required for these purposes if a large number of neighbourhood treatment plants is to be installed.

2.0 EXISTING SYSTEMS IN WESTERN AUSTRALIA

The Perth metropolitan sewerage system is made up of eleven wastewater treatment plants catering for an average flow of 230 ML/d (1997/98). Three large plants, one with only primary treatment and two with secondary treatment cater for 95% of the metropolitan flows. There are two medium sized plants and six small (local neighbourhood) plants.

In the country regions of Western Australia there are a large number of separate communities served by over 80 wastewater treatment plants in about 70 country towns. Many are pond systems which require large areas of land, which is generally available in the country regions, and are relatively cheap to run. However, in the country regions, the cost for the sewerage systems exceeds the revenue raised and hence must be subsidised by the revenue raised in the metropolitan area. This is shown in Table 2.1.

Table 2.1 Comparison of Metropolitan and Country costs for wastewater treatment (developer contributions not included)

ITEM	METROPOLITAN		COUNTRY	
	Total for 1997/98	Unit Cost c/m^3	Total for 1997/98	Unit Cost c/m^3
CAPITAL				
Interest	$ 22 930 000	46	$ 10 030 000	73
Depreciation	$ 64 380 000	63	$ 17 710 000	66
Salaries/Admin etc	$ 11 030 000	15	$ 4 210 000	21
OPERATING				
Direct Operation and Maintenance	$ 34 340 000	31	$ 10 370 000	51
Salaries/Admin etc	$ 12 000 000	16	$ 3 620 000	23
OTHER (incl. gov levy)	$ 12 420 000	11	$ 1 070 000	6
TOTAL	**$ 157 100 000**	**$1.82**	**$ 47 010 000**	**$2.41**
ANNUAL REVENUE RAISED (Largely Residential and Non-residential Rates and Charges)	$ 227 740 000	$ 2.24 /m^3	$ 32 070 000	$ 1.70 /m^3

3.0 OPERATION AND MAINTENANCE COSTS

Operating costs for wastewater treatment plants include the cost for power, chemicals, labour, maintenance, monitoring, supervision, administration, licensing and reporting. The small wastewater treatment plants that the Water Corporation owns and operates have high unit costs for operation and maintenance due to higher Direct Operating and Maintenance costs. This can be seen in Table 3.1.

Table 3.1 Approximate Unit Costs for some of the Current Water Corporation Wastewater Treatment Plants.

WWTP	Current Capacity (ep)	Capital Cost ($/ep)	Operation and Maintenance Cost ($/ep/yr) {c/m^3}
Small WWTP's			
Australind	3 750	270	40 {55}
Bridgetown	2 250	480	55 {75}
Bunbury #2	27 000	190	26 {36}
Busselton	22 500	200	28 {38}
Caddadup	3 000	400	47 {64}
Donnybrook	2 000	520	68 {86}
Dunsborough	7 500	270	31 {43}
Eaton	5 000	310	34 {47}
Pemberton	1 000	800	102 {140}
Metropolitan WWTP's			
Beenyup	500 000	250	14.6 {20}
Subiaco	300 000	270	16.1 {22}
Woodman Point (with proposed secondary treatment)	625 000	290	18.3 {25}

The Water Corporation has found that in general, for each tenfold increase in the size of the plant, the unit Direct Operating and Maintenance costs halve. This is shown in Table 3.2.

Table 3.2 Correlation between WWTP size and unit cost for Direct Operation and Maintenance

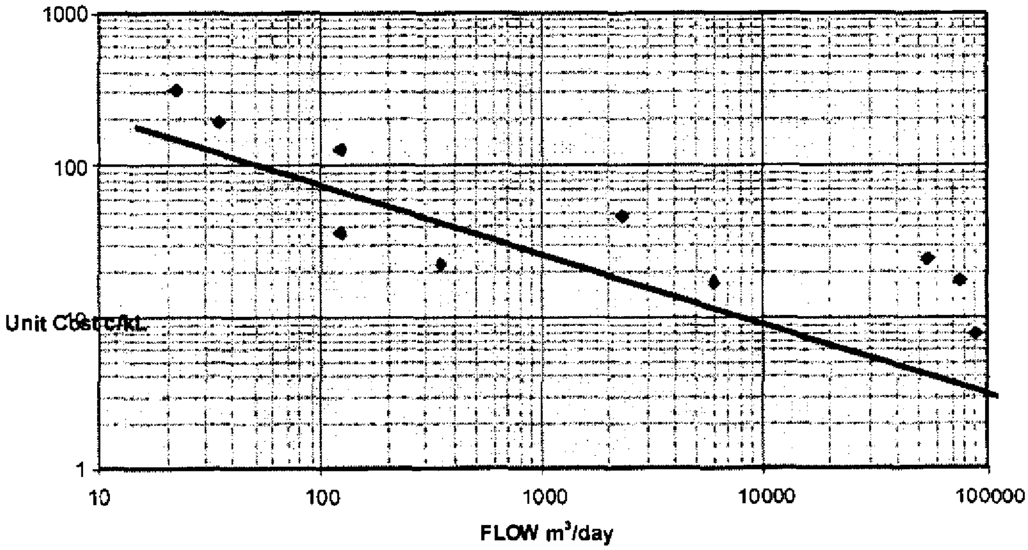
Plant Flow (ML/day)	Plant Population	Unit Cost (c/m^3)
0.05	280	160
.5	2 800	80
5	28 000	40
50	280 000	20

Extract from Wastewater 2040 Strategy for Perth Region

The operation and maintenance expenditure of a neighbourhood treatment plant serving 10 000 people would be 70 c/m^3 giving a total cost per year for 120 plants of $ 55.2 M, compared to that of a 400 000 ep centralised plant of about 20 c/m^3, and an annual operating cost of $17.5 M for three major treatment plants.

The graph below shows the Direct Operating and Maintenance Costs for metropolitan plants, all expressed in 1997/98 dollars.

DIRECT OPERATING AND MAINTENANCE COSTS - Metropolitan Secondary Treatment Plants



Some of the reasons why neighbourhood treatment plants incur a much higher unit cost for operation and maintenance are outlined further below.

3.1 Flexibility

The larger, centralised treatment plants tend to be more flexible than smaller neighbourhood treatment plants. They are generally built in modules that are designed and built as required. This may be in response to growth in wastewater flows or as a result of more stringent effluent and sludge disposal requirements. It could also be as a result of a demand for effluent for reuse. This type of facility allows the latest appropriate technologies to be incorporated in each new stage or upgrade.

The processes at Perth's three large plants use recent technology. Aspects of each plant are being updated on a regular basis. The egg shaped digesters at the Woodman Point

Wastewater Treatment Plant and the oil from sludge technology being installed at the Subiaco Wastewater Treatment Plant are examples of new and better technology being employed to improve the treatment process. Smaller plants tend to be built and operated unaltered for life.

Larger systems also have the buffering capacity to cope with changes in inflow characteristics either in terms of quality or quantity. Smaller systems can be more problematic because they may not always be able to cope with unusual events. Larger wastewater treatment plants have the capacity to cope with industrial wastes due to the high dilution with domestic wastewater in the sewerage reticulation system. Industrial wastes going into smaller treatment plants are likely to upset the biological processes. If small neighbourhood wastewater treatment plants were to be used throughout the metropolitan area, tighter industrial waste controls would be necessary.

3.2 Monitoring

Monitoring of both the process and the receiving environment is greater for larger plants. Perth's large treatment plants have automatic samplers which sample the incoming wastewater and the treated effluent continuously. One person is employed full time at each of these plants to monitor the processes. Analyses and checks are carried out daily. Because personnel are on site 24 hours a day, problems are often picked up by visual inspection. There is also a very intensive monitoring programme for monitoring the receiving environment. Over the last 2 years, $ 1M has been spent on the Perth Coastal Water Study and the Perth Longterm Ocean Outlet Monitoring (PLOOM) programme.

Smaller plants are generally only visited once a week and monitoring is carried out approximately weekly. Samples are collected manually from smaller treatment plants, hence are not as representative as those collected in the larger wastewater treatment plants by the automatic samplers. Because the sites are not manned continuously, there is less opportunity for problems to be identified by visual inspection of the processes. To sample and monitor 120 neighbourhood wastewater treatment plants, would require a huge resource and a lot of time would be lost travelling between the sites.

3.3 Licensing

All wastewater treatment plants with a flow greater than 100 m^3/day are required to be licensed by the Department of Environmental Protection. These licences are made up of two components. The first is a premises fee of $1250 which is applicable to all licensed sites and payable annually. Thus if Perth were to have 120 neighbourhood treatment systems, the premises fee alone would be $150 000 /yr. A much lower cost is incurred by having only three premises, with the total fee for the three centralised treatment plant option would being only $3 750/yr.

The second component of the fee is a charge for the pollutants put out into the environment. A charge is incurred for each kilogram of the various pollutants discharged annually. Assuming the quality of effluent achieved by neighbourhood treatment plants is similar to that achieved by the centralised treatment plants, this component of the fee would be similar. However, a Best Practice Environmental Licence waives the contaminant fees for the disposed effluent. Over the next few years, the three major metropolitan wastewater treatment plants will be moving towards being accredited with these types of licences. Obtaining these types of licences for numerous neighbourhood treatment plants would not be possible, due to the high level of administration requirements for Best Practice Environmental Licences and the associated quality systems.

3.4 Automation and manning

Large wastewater treatment plants tend to be more automated and are more closely monitored by well trained people. The two major secondary treatment plants in Perth, Beenyup and Subiaco are manned 24 hours a day seven days a week. This means that any problems that occur can be dealt with immediately, minimising down time of processes. Smaller plants tend to only be visited periodically during the week to do regular maintenance or to collect samples. The risk is therefore higher in smaller plants of faults not being identified immediately and thus threatening the stability of the quality.

The number of operators required to look after 120 neighbourhood wastewater treatment plants would be more than the total number of operators currently employed at the three major wastewater treatment plants. Two operators would be needed to 2 of the neighbourhood systems, this would mean 120 operators would be required in the metropolitan area. A huge amount of time would be lost in travelling between sites. Currently, the Water Corporation employs 34 Operators and three Plant Superintendents. At least 10 Plant Superintendents would be required to supervise 120 neighbourhood treatment plants.

3.5 Transport

Transport of chemicals and other consumables to over a hundred neighbourhood treatment plants would prove to be more difficult to administer and control than it currently is to the three major wastewater treatment plants. The level of heavy traffic in the suburbs would increase substantially.

3.6 Administration

An increase in the number of treatment plants results in an increase in the amount of administration required to run the facilities regardless of the size of the installations. Licences for each site are required by the Department of Environmental Protection and Department of Minerals and Energy if chemicals are stored on site. Permits may be required from the Water and Rivers Commission, depending on the method of disposal used. The Department of Environmental Protection Licences require an annual report to be produced and submitted. Each site would also require contracts to be in place for biosolids cartage and effluent reuse agreements would need to be in place if the treated effluent was to be reused.

4.0 ON SITE TREATMENT AND DISPOSAL

The alternative to neighbourhood treatment plants and large centralised systems is on site treatment and disposal methods. These include septic tank and soil absorption systems, modified septic tank systems such as Ecomax™, Aerobic Treatment Units and alternative technology such as composting toilets and greywater reuse.

Septic tanks have been deemed environmentally unacceptable in the Perth Metropolitan area and are slowly being phased out with the Infill Sewerage program.
There is an option in country areas for a system whereby septic tank waste from a number of properties is collected via a network of pipes and conveyed to a neighbourhood wastewater treatment plant. This eliminates the need for primary treatment in the neighbourhood treatment plant. This method obviously has the same land constraints and operation and maintenance difficulties as the neighbourhood treatment plant systems discussed in this paper and would not be feasible for the metropolitan area.

Modified septic tank systems, such as the Ecomax™ technology are more environmentally friendly than the standard septic tank system and produce a higher quality effluent but have a much higher capital cost, ie $6000 as opposed to about $3000 for a standard septic tank. They also have a higher annual operating and maintenance cost, as not only do they require regular sludge pump outs but also require periodic soil replacement. With the shift towards higher density living, the opportunity for these types of on-site treatment systems is decreasing.

Similarly, more advanced technology such as Aerobic Treatment Units have a higher capital cost again as well as higher operating and maintenance costs. They require maintenance contracts to be in place and also require power to operate. They obligate diligent operation by the owners to minimise health and environmental risks. This poses a problem when the property is sold as the new owners must also be committed to continuing a high standard of operation and maintenance on the unit.

Equally, alternative systems such as composting toilets and the on site reuse of greywater can be a good solution to the domestic waste problem in remote locations but is not suited to the metropolitan area. These systems also require diligent operation and maintenance by owners and thus it is unlikely that they will gain large scale acceptance in the community . "Any alternative management method must be as transparent as practical to users, allowing them to pay a service fee and then flush and forget it, as they do when using a conventional centralised system." (Venhuizen 1997)

5.0 OTHER CONSIDERATIONS

5.1 Environmental Impacts

If smaller neighbourhood wastewater treatment plants were to be scattered throughout the metropolitan area, there would obviously be more point sources of odour. Although with an adequate buffer area around the plant, odours would be minimised for sensitive land uses, there is still a risk of odour emission due to occasional unavoidable plant process failures. Also, it has proven difficult in the past to keep sensitive land uses out of buffer areas and it is not always possible for the Water Corporation to purchase the land required for buffer areas.

The primary environmental impact however would be the disposal or reuse of the treated effluent. As most of the neighbourhood wastewater treatment plant catchment areas would import most of their water through the metropolitan scheme water system, disposing of this water within the catchment would upset the local water balance.

Whether the treated wastewater within a neighbourhood catchment is disposed to the ground for aquifer recharge or reused as irrigation for public open space, the nutrient balance of the system will also be altered to some extent. There will be more nutrients in the system, resulting in more biomass produced within the catchment, increased nutrients in the soil or runoff to nearby waterways.

Similarly, the disposal of effluent to the ocean is not without impact on the environment. However, the lastest report from the Water Corporation's Perth Long Term Ocean Outlet Monitoring (PLOOM) program, confirms that the three ocean outfalls discharging treated wastewater into the sea off Perth are not having an adverse impact on the surrounding waters. Nor are there any early warning signs that suggest these waters will be harmed by increases in the wastewater flows in the next decade, or from the proposed upgrade of facilities at the Woodman Point Wastewater Treatment Plant.

An advantage of disposing of treated effluent to the ocean is that the disposal route is available year round. Often land disposal methods are only suitable during the summer months. In the winter treated effluent needs to be stored, so provisions would need to be made, for smaller wastewater treatment plants, for back up storage systems. This would impact on capital costs and increase the land area required.

5.2 Health Impacts

If water is extracted from bores within the catchment, disposing of effluent to the ground may pose health risks if that water is to be used as part of the potable water supply. With centralised treatment plants and disposal of treated effluent to the ocean there is no risk that treated effluent will mix with a potential potable groundwater supply.

A large portion of the Swan Coastal Plain is designated as either Class I, II or III Underground Water Pollution Control Areas (UWPCA's). These areas are managed to protect the groundwater from contamination, thus protecting the health of the general public. Effluent disposal to land in these areas would not be an option. Wastewater treatment plants are not permitted to be constructed on a Class I or II UWPCA's regardless of the method of effluent disposal.

On site wastewater treatment and greywater reuse, pose an even higher risk to the health of the public than both neighbourhood treatment plants and centralised treatment plants. The level of pathogens from septic tank wastes is not monitored. If not properly maintained, septic tanks can saturate the upper layers of the soil profile, bringing the waste into contact with humans. Likewise, aerobic treatment units are monitored infrequently and the potential for contact between humans and human waste is much higher than in a system whereby the wastes are transported away via a sewer reticulation system.

Composting toilets also increase the health risk to humans as the probability of contact between human wastes and humans is increased. Greywater can contain high levels of thermotolerant coliforms. If used for irrigation, there is a risk of pathogen contamination to humans. The risk is not limited to the owners and operators of these systems, but can be transferred to residents down gradient of these systems.

5.3 Social Impacts

The laying of main sewers can have a major social impact, and during the period of construction can severely disrupt peoples lives. This is offset however by the fact that sewers have a life in excess of fifty years, which minimises the social impact. Ocean outlets may also have an adverse social impact, particularly if not discharging a high quality treated effluent into deep waters well offshore. The general public now demand that beaches remain as near pristine as possible.

On site treatment and disposal causes minimal social impact, except for the loss of amenity value for parts of the garden which are used for effluent disposal.

There is no doubt that the greatest social impacts are created by wastewater treatment plants and effluent disposal sites. Whilst it is generally recognised that wastewater facilities are essential, the general view is that even when wastewater treatment plants are provided with an adequate buffer, they are highly undesireable in urban areas. For this reason, the construction of neighbourhood treatment plants in major cities is generally not adopted anywhere in the world.

CONCLUSION

When a sewerage system is in the planning phase, the major considerations should always be;

- protection of public health,
- protection of the environment,
- least cost, and
- least social impact.

From a public health point of view, greywater reuse is almost certainly the highest health risk, followed by on site treatment and disposal (particularly septic tanks), with the provision of a sewerage system and wastewater treatment plant presenting the least risk.

The same order applies when considering protection of the environment. In Perth the government's sewerage policy clearly identifies a preference for the provision of a sewerage system and a wastewater treatment plant. For the urban area of Perth the choice then is between neighbourhood plants and centralised systems. However, Perth will probably never be fully sewered and on site treatment and disposal will always have an important role to play. Again, whilst the concept of urban villages in Perth is worth developing, it is not considered essential to incorporate neighbourhood wastewater treatment plants for a scheme to succeed. Sewer mining is always a possibility to provide second class water, plus reuse of drainage water within the urban village would also supply a good quantity of second class water.

The main criteria in deciding between neighbourhood plants and centralised treatment plants will therefore be cost and social impact. Operating experience over many years clearly demonstrates that centralised treatment plants in Perth have a lower present value than neighbourhood plants. As the Water Corporation does not apply differential rating in Perth it will be very difficult for neighbourhood plants to be built even if the developer pays for the construction, unless a system can be developed to cover the shortfall in revenue.

It is known that centralised wastewater treatment plants have a greater social impact when main sewers are being laid. However this only occurs once every 40 or 50 years. Alternatively, the social impact of building 120 neighbourhood plants and effluent storage and disposal systems in Perth, together with the provision of buffer areas would have far greater social consequences.

Ocean disposal of treated wastewater is not favoured by many people. However the ocean is the original source of all our drinking water, and to return reclaimed water to the ocean can be seen as being part of the natural water cycle. Perth's soils do little to improve water quality, whilst the ocean will dilute, aerate and disinfect. Land disposal in Perth is difficult in winter and is not permitted in large parts of the metropolitan area.

At present the Water Corporation considers that suitably treated wastewater can be safely, effectively and economically treated in centralised wastewater treatment plants, before being returned to the environment by land and ocean disposal, with a portion being used as reclaimed water. However, smaller neighbourhood wastewater treatment plants do have a role to play in allowing new and innovative technology to be trialed and will be constructed in a few developments within the Perth metropolitan area.

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ON-SITE AEROBIC SYSTEMS

AEROBIC TREATMENT UNITS - AN ALTERNATIVE ON-SITE WASTEWATER TREATMENT SYSTEM

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ABSTRACT

Aerobic treatment units (ATU's) were advanced in Australia as an alternative on-site wastewater system. Poor design features, failure to meet discharge standards and lack of adequate maintenance has seriously questioned the suitability of these units. Western Australia developed specifications on design and performance and subjected units to a compliance test to meet BOD:SS 20:30; thermotolerent coliform count <10/100L and free chlorine residual <0.5mg/L. Quarterly maintenance is required by legislation.

Correctly designed and maintained ATU's can produce a reliably high quality wastewater. In association with proper disposal methods can provide effective reuse of the wastewater and is an alternative system which can be used in situations where conventional systems are not suitable.

KEYWORDS

Aerobic treatment units, on-site wastewater disposal, septic tanks, maintenance.

INTRODUCTION

Western Australia has relied on the septic tank - soil absorption system as the method of providing on-site water treatment and disposal where reticulated sewerage systems are not available.

Many of these systems were in the past under-designed consequently failures occurred presenting potential public health and environmental problems. Furthermore, systems were being installed where site and soil conditions were far from satisfactory for this form of conventional on-site disposal system.

The introduction of the aerobic treatment units (ATU's) [now referred to in Australia as aerated wastewater treatment systems (AWTS)] as a domestic wastewater system occurred in New South Wales in the early 1980's. These systems were primarily introduced to overcome failing septic tank systems that relied on regular pump-outs.

The benefits advanced for this new system were that a higher quality wastewater was produced which with chlorination enabled reuse of the wastewater by spray irrigation to garden areas. However, design features within the unit, poor installation practices and lack of maintenance have resulted in on-going concerns about the suitability of these units.

The Health Department of Western Australia is responsible for the approval of all on-site wastewater systems used in Western Australia. As a result of the evidence gained from the failure of ATU's in other states of Australia, the Health Department developed a specification on the design and performance criteria for ATU's. It is to be noted that a joint Australia/New Zealand standard is now being developed for these units based on experience from Western Australia.

ATU's were approved for use in Western Australia in 1991. At the present time four units are approved.

SYSTEM DESIGN

The ATU is designed primarily as a small domestic sewage treatment plant for 8-10 persons. The unit consists of primary treatment followed by accelerated aerobic breakdown, which is achieved through use of pumps/fans/air blowers and contact media. This is followed by settlement and chlorination. The final wastewater is pumped to an above ground irrigation area where it is dispersed through low volume sprays, or by a dripper system at or below ground level.
In developing the specification the emphasis was to ensure the unit was of robust design so that it could withstand overloading which could impact on the treatment process. The minimum design capacity was set at 8 persons with a maximum of 10 persons based on 180L/capita. Important design features include:

- addition of fixed media;
- separate baffled chlorine contact chamber;
- 2 days storage above normal operating level; and
- sloped base (55□) to clarification chamber.

Specific unit design features were:

HYDRAULIC LOADS

• average daily per capita flows	180L
• maximum per capita flow during any two hourly period	60L
• maximum flow in any 30 minute period	300L

BIOLOGICAL LOADS

• average daily per capita 5 day BOD	60g
• average daily per capita total suspended solids	60g
• average daily per capita total nitrogen	15g
• average daily per capita total phosphorus	2.5g

APPROVAL AND TESTING

To gain approval to manufacture ATU's detailed plans and specifications had to be prepared in regard to:

- hydraulic and biological load
- nutrient removal (if included in process)
- materials and construction
- mechanical and electrical equipment
- noise emission
- irrigation methods
- maintenance
- warranty.

Once plans were given desk top approval, the unit had to undergo compliance testing to meet the following criteria:

- BOD_5 — >20mg/L
- Suspended solids — >30mg/L
- Thermotolerant coliforms — >10/100mL
- Free chlorine residual — <0.5mg/L

Units were subjected to a minimum testing program (2 months) at a wastewater treatment plant where they were subjected to a loading of primary effluent at 1800 litres over 15 hours each day until the unit was ready for compliance testing.

Compliance testing was of four days duration, and performed under the supervision of the Health Department of Western Australia. Units were loaded at maximum flow capacity of 60L/capita for two hours and five samples taken at half hourly intervals for BOD_5, SS and thermotolerent coliforms. Units were then loaded at 300L for 30 minutes to check free chlorine residual.

The wastewater quality during compliance testing had to meet the following:

- 90% of BOD_5 less than or equal to 20mg/L with no sample greater than 30mg/L;
- 90% of total suspended solids less than or equal to 30mg/L with no sample greater than 45mg/L;
- free residual chlorine concentration in all samples greater than or equal to 0.5mg/L; and
- thermotolerent coliforms: median value not to exceed 10 organisms per 100mL, with four out of five samples containing less than 20 organisms per 100mL.

FIELD TESTING

ANNUAL SURVEYS, 1992-1996

Since 1992, sixteen (16) units have been tested approximately every twelve months to determine on-going performance and compliance with discharge criteria. In 1995, a further 5 units were included.

The parameters tested for each unit and the sampling points were:

- BOD:SS - clarifier
- pH - disinfection chamber
- Temperature - aerobic chamber
- Dissolved Oxygen - aerobic chamber
- Free Chlorine - disinfection chamber
- Thermotolerant Coliform Count - disinfection chamber

In addition, the clarity of the effluent and the condition of the irrigation areas were observed.

The results of the testing are shown in Appendix 1, Table 1. The overall compliance for the five year period against the performance criteria is summarised as:

Parameter	No of Samples	Compliance
Free chlorine <0.5mg/L	37	54%
Thermotolerant Coliforms <10/100ml	54	76%
Dissolved oxygen >2mg/L	68	97%
BOD:SS	48	68%

In 1995 a further 52 units were tested as a one-off survey. These results were similar to the above.

The most significant parameter to be met when relying on above ground spray irrigation is the thermotolerent coliform count. When a unit is correctly maintained with sufficient supply of chlorine tablets excellent results are achieved. The reasons for poor results are:

- tablets jammed in the chlorination dispensers, preventing contact with the effluent for adequate disinfection;
- tablets worn unevenly so contact is minimal when the base tablet is worn down;
- no chlorine tablets - several units were due, or overdue, for maintenance; or
- no baffle in chlorine chamber.

NUTRIENT REMOVAL

In environmentally sensitive areas the use of on-site disposal has been restricted to systems which limit the amount of phosphorus discharged from the site. ATU's can be used for this purpose. One unit approved in Western Australia has built within the unit an alum dosing system that controls the phosphorus. The discharge criteria is set at less than 1mg/L phosphorus. The unit was subjected to testing to ensure compliance with this requirement.

Another option approved is the application of a mixture of red mud material and sand to the irrigation area. This only applies to spray irrigation areas and requires the application of $30m^3$

of red mud material over 150m^2 of irrigation area. The red mud material is the residue from alumina processing and has been proven to have good phosphorus adsorption qualities.

IRRIGATION AREA

Spray irrigation has been the most widely accepted method for the disposal of wastewater from ATU's. In Western Australia the minimum spray irrigation area for a single dwelling is 150m^2. The irrigation area must be:

- set up as a dedicated area, eg: garden area (not lawned area or used to grow vegetable crops);
- set back from human traffic areas;
- sign posted (reuse of wastewater);
- maintained to ensure complete distribution of wastewater over area.

Experience has shown that spray irrigation is not the method which should be supported particularly on small allotments. Spray irrigation does bring the home owner and animals into possible contact with the wastewater. Failures occur with the chlorination system resulting in the application of the wastewater with high thermotolerant coliform counts and as such is a potential public health risk. Furthermore many irrigation areas are poorly maintained and lack adequate plant cover. The use of dripper irrigation is to be recommended as not only does this virtually eliminate the public health risk but uptake by plants is more effective by this method.

MAINTENANCE

Maintenance is fundamental for the satisfactory performance of the ATU. In Western Australia it is a legal requirement for the owner of a premises on which an ATU is installed to have a maintenance contract in place. Before approval is given to install a unit the maintenance contract must be lodged with the application.

Maintenance of the ATU must be undertaken every three months and this can only be carried out by an authorised person. A copy of the maintenance report must be submitted to the local government and Health Department of Western Australia. The maintenance report includes details such as:

- condition of the wastewater in various chambers;
- sludge/skimmer return;
- aeration;
- chlorine reading;
- supply of chlorine tablets;
- irrigation area;
- replacements/repairs undertaken.

In the past there has been no formal training for authorised persons undertaking maintenance. It is now proposed to introduce a formal training program.

Home owners also need to be fully informed on the operation and maintenance of such units and their responsibilities in managing the irrigation areas and the limits to its use.

CONCLUSION

The development of specifications for the design and performance of ATU's, as adopted in Western Australia, together with testing of units for compliance, has demonstrated that ATU's can meet performance criteria. However, for reliability in the long term and to prevent potential public health risks the units must be maintained on a regular basis. Maintenance should be mandatory under legislation and carried out by suitably trained personnel. The use of above ground spray irrigation does increase the public health risk due to inadequate maintenance or system failure. The use of a below ground dripper system overcomes this risk and should become the method of disposal.

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APPENDIX 1

Table 1: Annual Survey of Aerobic Treatment Units (1992 - 1996)

Unit	Time of Operation	Date of Visit	No. of Occup.	BOD:SS mg/L BOD	SS	pH	Temp °C	Diss. O_2 mg/L	Free Chlorine mg/L	TCC per 100mL
1	4 years 10 months	7/92	2	5	35	6.8	20.3	6.8	0.2	<5
		11/93		10	20	6.8	20.5	6.9	4.0	<5
		3/95		<5	5	6.8	24.0	7.8	1.0	0
		8/96	2	<5	5	3.5	16.7	6.9	0.29	0
2	5 years 3 months	7/92	4	55 (15)	30 (40)	7.0	20.8	4.0	0.2	<5
		11/93		25	20	7.2	25.0	5.0	1.5	<5
		3/95		15	20	7.4	27.0	5.7	0.4	0
		8/96	4	40 (15)	280 (5)	6.7	18.1	5.0	0.5	<10
3	5 years 4 months	7/92	5	15 (20)	20 (15)	6.8	17.7	6.0	<0.2	23,000
		11/93		70	85	-	22.6	<1.0	0.4	<5
		3/95		5	<5	7.3	25.0	5.5	2.5	0
		8/96	6	10	25	7.3	16.8	5.4	0.3	<10
4	4 years 11 months	7/92	5	(15)	(45)	5.2	18.7	6.2	0.2	15
		11/93		20	20	7.2	23.0	5.3	1.0	<5
		2/95		<5	10	6.7	27.0	5.9	0	>10,000
		8/96	3	20	60	5.3	17.8	5.9	0.89	<10
5	4 years 6 months	7/92	2	15 (5)	15 (10)	6.8	17.8	6.3	0.4	5
		11/93		5	10	7.4	22.6	9.8	0.4	<5
		2/95		<5	20	7.5	27.0	6.0	1.5	<10
		8/96	1	10	15	4.1	16.1	8.2	5.3	<10

NOTE: 1. TCC = Thermotolerant coliforms
2. BOD:SS - results are from clarification chamber, those shown in brackets are from disinfection chamber.

Unit	Time of Operation	Date of Visit	No. of Occup.	BOD:SS mg/L BOD	SS	pH	Temp °C	Diss. O_2 mg/L	Free Chlorine mg/L	TCC per 100mL
6	5 years 11 months	7/92	5	10 (15)	5 (10)	7.2	*	*	0.5	<5
		10/93		<5	10	7.0	20.5	9.6	4.0	5
		3/95		<5	10	6.8	26.0	5.9	4.0	0
		8/96	2	15	15	6.0	17.9	6.2	2.4	<10
7	5 years	7/92	4	(5)	(15)	6.7	-	6.7	1.0	5
		10/93		<5	10	7.2	16.1	5.6	0	<5
		2/95		10	10	7.5	25.0	4.0	1.5	<10
		8/96	5	20	20	7.2	16.0	4.1	0.07	0
8	4 years 10 months	7/92	4	5 (30)	15 (15)	7.4	18.6	5.8	>4	<10
		10/93		15	15	7.4	19.1	2.9	1.5	<5
		2/95		10	5	7.3	27.0	3.7	1.0	<10
		8/96		120 (20)	940 (<5)	7.1	17.9	3.9	0.22	0
9	5 years	7/92	6	(15)	(30)	6.8	-	4.9	0.5	20
		10/93		20	50	7.4	15.7	5.4	0	1900
		2/95		-	-	-	-	-	-	-
		8/96	4	30	30	7.5	14.7	8.2	0.06	0
10	5 years 2 months	7/92	4	(20)	(25)	6.8	-	4.2	0.8	<5
		10/93		15	25	7.4	19.7	6.5	1.0	5
		2/95		25	5	7.6	26.0	2.4	1.5	<10
		8/96	5	40	65	7.3	18.0	3.6	0.13	0

Table 1: Annual Survey of Aerobic Treatment Units (1992 - 1996)

Unit	Time of Operation	Date of Visit	No. of Occup.	BOD:SS mg/L BOD	SS	pH	Temp °C	Diss. O_2 mg/L	Free Chlorine mg/L	TCC per 100mL
11	4 years 6 months	7/92	4	(10)	(10)	7.2	-	5.1	2.0	<5
		10/93		10	10	7.2	17.0	5.8	1.5	<5
		2/95		<5	5	6.9	24.0	3.5	4.0	<10
		8/96	4	15	5	6.2	15.6	4.5	3.7	0
12	6 years 4 months	7/92	4	30 (5)	40 (25)	6.8	20.3	6.8	1.0	<5
		11/93		30	25	6.8	24.4	3.5	1.5	<5
		3/95		10	10	6.8	24.0	4.2	0	7,900
		8/96	4	40	35	7.4	19.1	3.7	0.19	>200
13	6 years 2 months	7/92	4	35	25	6.8	-	7.5	1.0	<5
		10/93		35	45	7.4	19.9	6.3	1.0	5
		2/95		15	10	7.4	25.0	3.7	0.4	<10
		8/96		80	150	7.5	18.4	2.9	0.08	0
14	10 years 3 months	7/92	4	15 (30)	40 (25)	6.8	22.5	5.3	0.4	460
		10/93		35	130	7.0	21.8	5.3	1.5	45
		2/95		<5	<5	7.4	25.0	4.6	0.6	<10
		8/96	4	15	25	7.6	22.4	2.2	0.07	0
15	4 years 4 months	7/92	2	(5)	(<5)	6.6	15.4	9.9	0.2	<10
		10/93		<5	<5	6.6	18.4	11.5	1.0	<5
		2/95		<5	<5	7.4	25.7	8.4	0.2	<10

Table 1: Annual Survey of Aerobic Treatment Units (1992 - 1996)

Unit	Time of Operation	Date of Visit	No. of Occup.	BOD:SS mg/L BOD	BOD:SS mg/L SS	pH	Temp °C	Diss. O_2 mg/L	Free Chlorine mg/L	TCC per 100mL
16	3 years 3 months									
		3/95		20	20	7.4	24.0	6.6	1.0	2
		8/96	2	20	15	7.7	13.3	0.1	0	164
17	3 years 4 months									
		3/95		10	25	7.8	28.0	3.5	0.2	>200
		8/96	4	5	10	7.2	19.5	3.2	0.66	<10
18	3 years 10 months									
		3/95		40	120	6.8	25.0	2.4	0.2	>10,000
		8/96	3	45	150	7.8	17.4	4.0	0.10	14,400
19	3 years 7 months									
		3/95		25	25	7.6	28.0	4.0	2.5	0
		8/96	3	20	10	7.5	19.1	0.4	0.25	<10
20	3 years 8 months									
		2/95		20	<5	7.4	26.0	6.4	0.2	330
		8/96	3	35	40	7.5	16.9	4.7	0.03	85,000
21	Approx. 5 years	7/92	-	(10)	(5)	6.6	16.7	9.0	<0.2	180
		-	-	-	-	-	-	-	-	-
		2/95	2	<5	<5	3.5	23.0	8.6	-	50
		8/96	1	<5	<5	5.1	15.9	8.6	-	1,020

Table 1: Annual Survey of Aerobic Treatment Units (1992 - 1996)

EFFECTIVENESS OF ALKALINE PRECIPITATOR FLY ASH AND SAND AMENDED WITH IT FOR PHOSPHORUS REMOVAL AND SEPTIC TANK EFFLUENT RENOVATION

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ABSTRACT

Septic tank effluent is customarily disposed of by soil infiltration. In sandy soils with minimal pollutant attenuation capabilities, such as found in Perth, the risk of ground and surface water pollution is high. This study investigated phosphorus removal capabilities of alkaline precipitator fly ash and the latter's use as a soil amender for effecting enhanced treatment of septic tank effluent.

Initial batch contacting tests using phosphate solutions indicated that alkaline precipitator fly ash possessed the highest P sorption capacity among a variety of materials studied in terms of its Langmuir and Freundlich sorption isotherm parameters. In order to gain a better indication of longer term P sorption capability, the test materials were repeatedly contacted with fresh phosphate solutions over 90 contacting cycles. Again, precipitator fly ash exhibited a higher P sorption capacity than the other materials.

In the next phase, infiltration studies were conducted using laboratory soil columns to assess the treatment effectiveness on septic tank effluent of Spearwood sand amended with alkaline precipitator fly ash at different levels. The columns were operated continuously for 12 weeks at a hydraulic loading rate of 4cm/day. Concentrations of COD, nitrogen, phosphorus, and selected metals were monitored in the effluent from all columns. Similar extent of COD reduction was achieved in all sand columns with or without precipitator fly ash. On the other hand, nitrogen and phosphorus removals were significantly enhanced by fly ash amendment of the sand. Increased levels of fly ash in the soil columns resulted in increased nitrogen and phosphorus removal. However, high application rates of fly ash caused clogging of the infiltration bed apparently due to the lower permeability. It is reasoned that 5 to 15% fly ash could be added to coarse sands to produce an infiltration bed which would result in a better quality effluent than can be obtained with untreated Spearwood sand alone.

KEYWORDS

Fly ash, soil amendment, soil infiltration, phosphorus removal, septic tank effluent.

INTRODUCTION

Septic tanks with associated soil infiltration systems are widely used for on-site domestic wastewater disposal in rural and isolated communities as well as in many unsewered urban localities, including parts of Perth, Western Australia. Where soil characteristics are suitable, the system helps attenuate many pollutants before they are carried into ground water (Reneau *et al.*, 1989). Poor phosphorus retention is often noted in sandy soils (Magdoff *et al.*, 1974; Whelan and Barrow, 1984; Pell and Nyberg, 1989) and in these situations, phosphorus entering ground water can subsequently cause eutrophication problems in streams and lakes.

Artificial infiltration beds are used when natural soil exhibits inadequate drainage or pollutant attenuation (Kristiansen, 1981; Reneau *et al.*, 1989; Gold *et al.*, 1992). Research has shown that enhanced P removal can result with infiltration beds employing sand mixed with one of several soil amenders. The soil amendments investigated include the addition of clay (Willman *et al.*, 1981), red mud (the fine fraction of the residue from bauxite refining) neutralised with gypsum (Ho *et al.*, 1992), iron oxides, steel wool (James *et al.*, 1992) and metallic iron particles (Wakatsuki *et al.*, 1993). The use of industrial residues like red mud and gypsum is attractive as it presents an avenue to beneficially use an otherwise troublesome waste material. Clearly, it would be desirable to find similar potential uses for other industrial solid wastes.

Fly ash is a waste product of power stations burning pulverised coal. The likely benefits and possible problems of using fly ash on agricultural land to improve physical and chemical properties of soil have been widely studied (Plank and Martens, 1973; Adriano *et al.*, 1980; Korcak and Kamper., 1993; Kukier *et al.*, 1994). Also, many researchers have examined effluent treatment with addition of fly ash to remove various pollutants. These have included the removal of COD (Eye and Basu, 1970), detergent (Bhargava *et al.*, 1974), suspended solids and dissolved metals (Gray *et al.*, 1988), organic impurities (Banerjee *et al.*, 1989), chrome dye (Gupta *et al.*, 1990), heavy metals (Panday *et al.*, 1985; Sen and De, 1987; Diamadopoulos *et al.*, 1993) and phosphorus (Gangoli and Thodos, 1973; Higgins *et al.*, 1976; Vinyard and Bates, 1979; Tsitouridou and Georgiou, 1988; Gray and Schwab, 1993). In contrast, little information exists on the use of fly ash as a soil amender for infiltration beds and its effectiveness on effluent purification, particularly with respect to the removal of nitrogen and phosphorus. Soil microbial activity is an important factor contributing to organics decomposition and nitrogen transformations during effluent infiltration. Several studies have indicated adverse effects on soil microbial activity from fly ash addition (Wong and Wong, 1986; Pichtel, 1990; Cervelli *et al.*, 1986; Pichtel and Hayes 1990). Finally, there has been some concern about possible release of various dissolved contaminants including acidic or alkaline solutes and heavy metal ions from fly ash (Dreesen *et al.*, 1977; Dudas, 1981; Hodgson *et al.*, 1982; Humenick *et al.*, 1983; Roy *et al.*, 1984; Wadge and Hutton, 1987).

In the present study, the phosphorus removal capability of an alkaline precipitator fly ash was examined and compared with a number of other materials. Initially P sorption by the test materials was investigated during short-term batch contacting tests. The results from these tests were represented and compared in terms of conventional adsorption isotherms. In the next phase, a more elaborate examination of P sorption characteristics was undertaken in which the adsorbent samples were allowed to react with fresh phosphate solutions over 90 contacting cycles. The data from short-term contacting studies were assessed against these results to gain insight into the usefulness of short term contacting data when evaluating alternative adsorbent materials. Finally, a series of continuous infiltration studies was undertaken in which septic tank effluent was applied to laboratory soil columns containing sand amended with various levels of alkaline precipitator fly ash. The general treatment efficiency of the infiltration beds was assessed particularly with respect to phosphorus and nitrogen removals and the release of metal ions into the treated effluent.

MATERIALS AND METHODS

Short term batch P sorption studies

Fresh (unweathered) alkaline precipitator fly ash used in this study was obtained from the Castle Peak power station in Hong Kong. In addition, phosphorus sorption characteristics of the following materials were ascertained for comparison purposes: a partially weathered sample of fly ash from the storage lagoon at the Castle Peak power station, a sample of red mud treated with 5% by weight gypsum from Alcoa of Australia in Western Australia (WA), Merribrook soil (a loamy sand) from Albany in WA and Spearwood sand (a coarse sand) from Perth in WA. All materials were air-dried and passed through a 2 mm sieve prior to their use in experiments.

Triplicate 5 g samples of the adsorbent materials were placed in 25 ml of 0.01M KCl solutions containing different amounts of P as KH_2PO_4. The suspensions were continually shaken at 25°C for 24 h. At the end of the 24 h period, the pH was measured, the suspension was centrifuged and the supernatant was analyzed for phosphorus using the ascorbic acid method (APHA, 1989). The phosphate sorption data were fitted to the Freundlich and Langmuir equations shown below:

$$X = kC^n \ (\textit{Freundlich isotherm}) \quad \text{and} \quad X = \frac{bX_mC}{1+bC} \ (\textit{Langmuir isotherm})$$

where X = specific P sorption by the material (mg P/kg adsorbent), C = residual phosphorus concentration in solution (mg P/L) and k, n , b and X_m are parameters of the isotherm equations.

P sorption by repeated contacting with fresh phosphate solutions

Triplicate 5 g samples of the adsorbent materials were placed in 50 mL centrifuge tubes. 25 mL of 0.01M KCl solution containing 500 μg P (20 mg/L) as KH_2PO_4 were added. The phosphorus concentration in the solution was similar to the average concentration in WA septic tank effluent (17.7 mg/L). The samples were shaken at 25°C for 24 hours. The tubes were then centrifuged, the supernatant was decanted and analyzed for unsorbed phosphorus. The centrifuged solids were resuspended in 25 ml fresh P solution and P sorption was allowed to occur for a further 24 h. The procedure was repeated 90 times for each material tested. The amounts of adsorbed P was calculated by adding up decreases in residual P concentration in the solutions through successive contacting cycles.

Septic tank effluent infiltration studies with soil columns containing sand amended with precipitator fly ash

For these studies soil infiltration media were prepared using Spearwood sand amended with 0, 5, 15 and 30% Castle peak precipitator fly ash. The infiltration experiments were carried out in PVC columns (2.5 cm id.) containing a base of 2 cm of washed glass beads overlaid by 30 cm of prepared soil medium. The surface of the medium was covered with 2 cm thick glass wool. Three replicate columns were employed for each test material. Septic tank effluent was collected from the Kwinana wastewater treatment plant. The average characteristics of the wastewater are shown in Table 1. 20 mL of septic tank effluent was applied daily to each column (hydraulic loading of 4 cm/day) for 12 weeks. Effluent samples were collected weekly for analysis. All column studies were conducted at 25°C.

Influent and effluent samples were collected, preserved and analysed in accordance with Standard Methods (APHA, 1989). COD was determined by titration with ferrous ammonium sulfate after closed reflux digestion with dichromate. Soluble COD was determined after filtration through glass fibre paper. Total suspended solids (TSS) was determined gravimetrically by filtration through glass fibre paper. NH_4^+-N was measured by the phenate method;

$NO_3^- + NO_2^- - N$ by the cadmium reduction method and total Kjeldahl nitrogen (TKN) by digestion with Kjeldahl tablets. $PO_4^{3-} - P$ was analysed by the ascorbic acid method and total phosphorus by sulfuric acid-nitric acid digestion followed by phosphate determination. Dissolved metal ion concentrations in the feed and effluent samples were determined by inductively coupled argon plasma spectrophotometry on samples filtered through 0.45μm filter paper. Analyses were done for Al, Ca, Fe, As, Cd, Cr, Mn, Pb, Se and Zn.

Table 1. Mean characteristics of septic tank effluent from Kwinana wastewater treatment plant. (*All values except pH are in mg/L.*)

Principal effluent parameters		*Soluble metal concentrations*	
PH	7.1	Al	< 0.05
Alkalinity, as CaCO3	270	Ca	18.1
COD (unfiltered)	122	Fe	0.07
COD (filtered)	64	As	< 0.05
Total suspended solids	130	Cd	< 0.05
Phosphate – P	17.6	Cr	< 0.05
Total – P	19	Mn	< 0.05
Ammoniacal - N	76.7	Pb	< 0.05
Total Kjeldahl - N	78.7	Se	0.11
Nitrate and nitrite - N	0.02	Zn	< 0.05

RESULTS AND DISCUSSION

Short term batch P sorption

The Langmuir or Freundlich isotherms have been commonly used to describe P sorption data (Stuanes, 1984; Singh and Gilkes, 1991; Porter and Sanchez, 1992; Agbenin and Tiessen, 1994). Details on the fitting of the batch P sorption data to the isotherm equations have been discussed elsewhere (Cheung *et al.*, 1994; Cheung, 1997). The experimental data for the studied materials fitted the isotherm equations well. The correlation coefficients for the least squares fits ranged from 0.97 to 0.99. The estimated values of the isotherm parameters are shown in Table 2.

The Freundlich coefficient, k generally showed a good correlation with the Langmuir sorption maximum, X_m. Although others (Holford, 1982; Singh and Gilkes, 1991) have also obtained similar comparisons, it should be noted that k and X_m do not actually reflect the same characteristics. The Freundlich k represents the amount of P sorbed when the solution concentration is unity. X_m, on the other hand represents the saturation level of sorbed P at high solution concentrations. Similarly, the parameters n and b are also not directly comparable. b in the Langmuir model measures the affinity of the sorbent for the solute (a high value of b means a high sorption level at low solution concentration). On the other hand n measures the extent of impact on sorption of a change in solution concentration from unity. (A high value for n implies a relatively large change in sorbed P when the solution concentration deviates from unity - either above or below it). The Freundlich parameter k and the Langmuir parameter b, both measure, in a sense, the effectiveness of P sorption at low P concentration in solution and so may be expected to show good correlation. The data in Table 2 demonstrates this to be the case. Similarly, the ratio of X_m to k can be thought to reflect the relativity of the extent of sorption at high and low solution P concentrations. This quantity may be expected to correlate well with the Freundlich parameter n. Again, the data in table 2 is in agreement with this expectation.

Table 2. Freundlich and Langmuir isotherm data for P sorption by various materials

Sorbent material	Freundlich isotherm Parameters		Langmuir isotherm Parameters	
	K (L/kg)	n	X_m (mg/kg)	b (L/mg)
Castle Peak precipitator fly ash	12300	0.01	13766	0.28
Red mud gypsum	498	0.58	5070	0.081
Castle Peak lagoon fly ash	113	0.58	3082	0.015
Merribrook loamy sand	396	0.19	1490	0.02
Spearwood sand	4.4	0.18	35	4.81

As indicated by the Freundlich 'k' and the Langmuir 'X_m' values, Castle Peak precipitator fly ash exhibited the highest P sorption capacity of the five materials tested. Based on these data the sorption capacity of Castle Peak lagoon fly ash would appear to be even greater than that of red mud with added gypsum, which is widely regarded as a good sorbent for phosphate removal. Alkaline fly ashes contain high levels of extractable calcium (Mattigod *et al.*, 1990) and their high P sorption capacity is likely related to this. The calcium would precipitate phosphate above pH 7. All three different species of calcium phosphate (dicalcium phosphate dihydrate, dicalcium phosphate anhydrous, and octacalcium phosphate) can precipitate if the total phosphorus concentration is higher than 4 mg/L and the Ca^{2+} activity is at least 200 mg/L (Stuanes, 1984). The very low value of n for the fly ash indicates that the sorption capacity is relatively independent of solution P concentration. Its relatively high affinity for P is also indicated by the high value of the Langmuir coefficient b.

Gypsum amended red mud exhibited the second highest P sorption capacity. The high value of its Freundlich parameter, n suggested strong dependence of sorbed P on solution P concentration. Castle Peak lagoon fly ash and Merribrook loamy sand also exhibited significant P sorption capacities, but less than those of precipitator fly ash and red mud gypsum. Spearwood sand showed only minimal capacity for P sorption which suggests that significant P attenuation is unlikely to occur during effluent infiltration through this sand.

P sorption on repeated contacting with fresh phosphate solutions

The P 'breakthrough' curves for the selected materials during repeated contacting with P solutions over 90 cycles are shown in Figure 1. Initially, added P was nearly completely sorbed by the materials. Subsequent contact cycles led to progressive reduction in P sorption from one cycle to the next and eventual P "breakthrough'. Over 1 mg/L residual P in the solution was found by the 10^{th} contacting cycle for all materials except for precipitator fly ash. The latter sorbed virtually all the added phosphorus until the 50^{th} cycle. P breakthrough concentration equal to that in the feed solution occurred with Spearwood sand after just 1 contact cycle and with the lagoon fly ash after 13 cycles.

For Merribrook soil, precipitator fly ash and red mud gypsum, even after the major P breakthrough, further P uptake continued at a steady rate (albeit a smaller proportion of the feed) throughout the experiment. This behaviour accords with the hypothesis of two distinct phases of P sorption – an initial large uptake lasting only a short time followed by further removal at significantly reduced levels, but sustained over longer periods - similar to that suggested by Barrow and Shaw, 1974 and Ryden *et al.*, 1977.

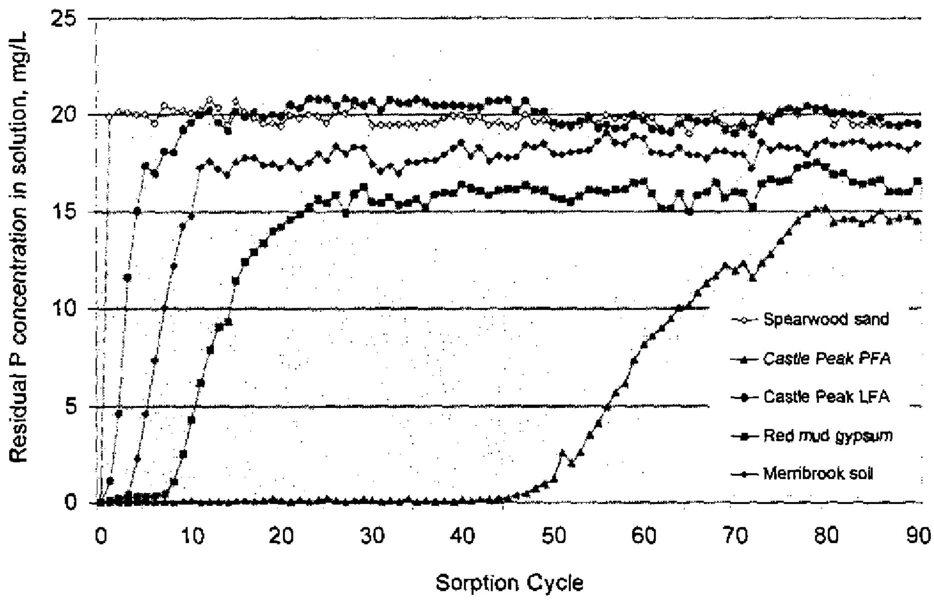

Figure 1. Breakthrough curves for P sorption by repeated contacting with fresh phosphate solutions

Table 3 compares the actual P sorption achieved over 90 contact cycles with the Langmuir sorption maximum, X_m. There is significant disparity among the different materials with respect to the actual P sorbed relative to their respective X_m values. The particular reasons for the differences are not clear, but the data do suggest that sorption isotherms based on short term contacting do not accurately mirror the real sorption characteristics of the materials. On the other hand, a high correlation ($R^2 = 0.963$) was obtained between the Langmuir sorption maxima (X_m) and the measured P sorption over 90 contact cycles. This suggests that while the Langmuir X_m value may not accurately estimate the long-term sorption capacity of materials, it could still be useful for comparison of alternative materials for infiltration media.

Table 3. Comparison of P uptake over 90 contact cycles with the Langmuir sorption maximum

Adsorbent material	P sorbed in 90 cycles (mg/kg)	Ratio of P sorbed to Langmuir sorption maximum, X_m (%)
Castle Peak precipitator fly ash	6790	49.3
Red mud gypsum	2810	55.5
Castle peak lagoon fly ash	275	8.93
Merribrook loamy sand	1480	99.6
Spearwood sand	111	320

Septic tank effluent infiltration studies with soil columns containing sand amended with precipitator fly ash

Phosphorus removal

The bulk of the total phosphorus (~95%) present in the septic tank effluent was in the form of phosphate (Table 1). Phosphorus leakage in the effluent occurred early in the unamended sand column (Figure 2). By week 7, the effluent concentration had risen to about 12 mg/L, equating to a removal efficiency of around 35%. Effluent P concentration progressively increased beyond this time, but at a noticeably slower rate.

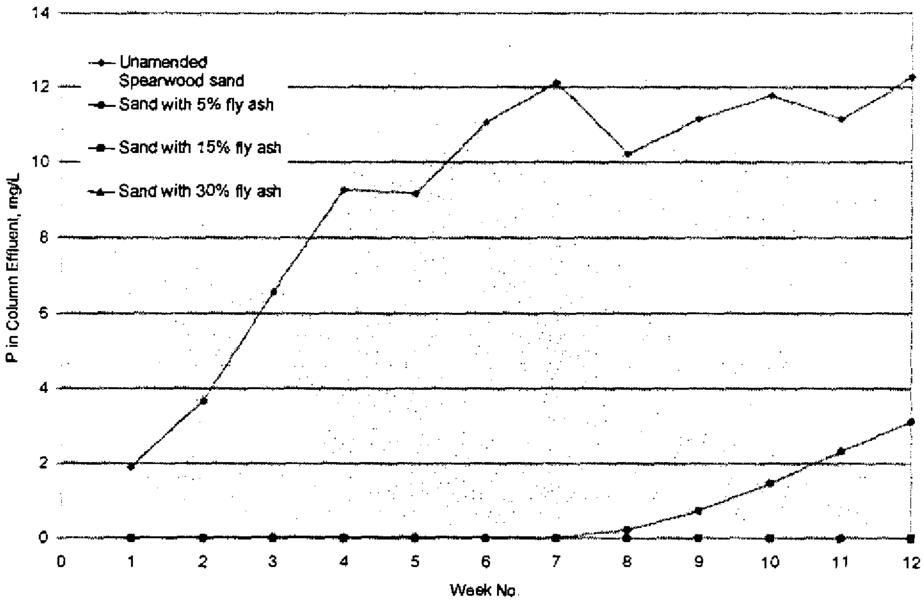

Figure 2. Phosphorus removal in soil columns treating septic tank effluent

Substantial improvement in P removal was noted with the amendment of the sand with precipitator fly ash. With 5% fly ash addition, no P leakage occurred in the effluent until the eighth week. From this point on, P concentration showed an apparent steady increase over time. With 15% and 30% fly ash amendments, no P was detected in the effluent up to the end of the experimental run. Cheung (1997) has shown that the capacity for P sorption by soils and other adsorbents are related to the occurrence of crystalline iron oxides and/or exchangeable calcium. Thus Spearwood sand which has low levels of crystalline iron oxides and exchangeable calcium is unable to retain much P from the treated effluent. Castle Peak precipitator fly ash, on the other hand, contains a high level of exchangeable calcium and only small amounts of crystalline iron oxide. The observed P removal thus would have been primarily due to precipitation with calcium ions.

It is interesting to speculate on the progress of P sorption by the amended sands beyond the period of the experiment. On the one hand, based on the results of Figure 1, a certain extent of P sorption would likely be maintained for some considerable period – corresponding to the 'second phase' of P sorption discussed earlier. On the other hand, a high concentration of calcium was measured in the amended column effluents, indicating that calcium was rapidly being leached from the columns. If P removal occurred mainly through precipitation by calcium ions, the leaching of calcium would curtail the capacity for further P retention by the infiltraion bed. Curiously, however, the columns with 30% fly ash showed significantly lower calcium leakage than columns containing 5 and 15% fly ash. Thus it would seem fair to conclude that the question of long term P removal by fly ash amended sand columns cannot be unequivocally answered without further research involving longer term experimental investigations.

Nitrogen transformations and removal

Ammonia nitrogen was the predominant form of N (approximately 97%) in the septic tank effluent. Mechanisms for ammonia nitrogen removal include volatilisation, sorption and cation exchange in the filter media, biological uptake for growth, and nitrification. The principal mechanisms would of course be dependent on the conditions in the infiltration media.

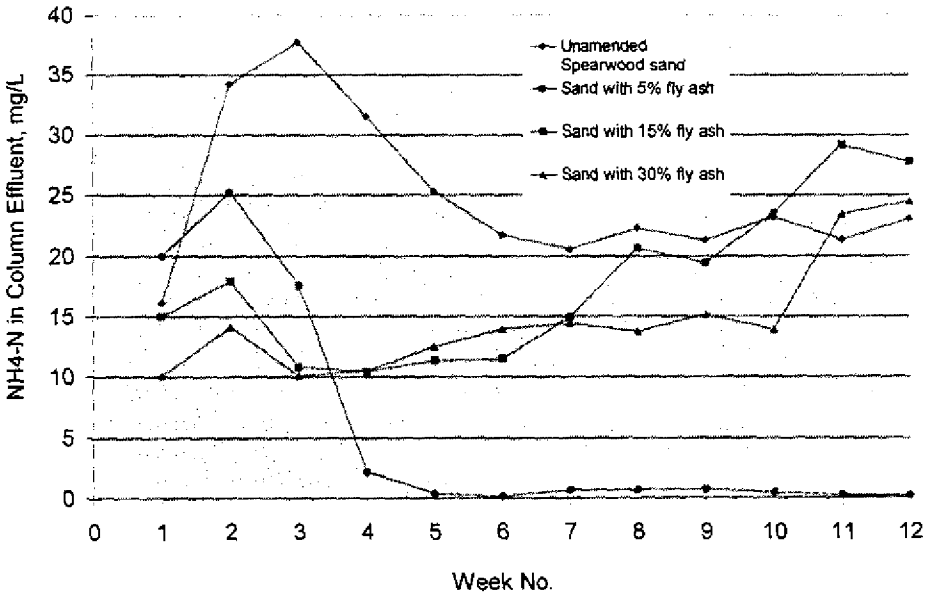

Figure 3. NH_4^+-N removals in soil columns treating septic tank effluent

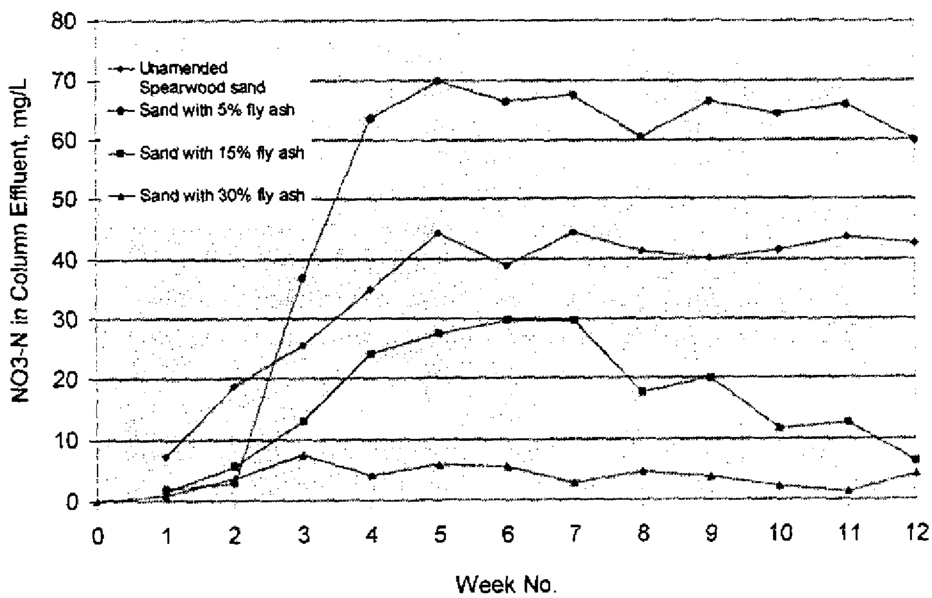

Figure 4. $NO_3^- + NO_2^- - N$ levels in soil columns treating septic tank effluent

Figures 3 and 4 show respectively the NH_4^+-N and $NO_3^- + NO_2^- - N$ nitrogen levels in the effluents from the various soil columns. Virtually no NH_4^+-N was encountered after week 4 in the effluent from the column containing 5% fly ash amended sand. The bulk of the NH_4^+-N removed in this column appeared as $NO_3^- + NO_2^- - N$ showing that this column was efficiently nitrifying virtually all of the feed nitrogen, but little of the feed nitrogen was actually being removed. In all other columns including the unamended sand column, effluent NH_4^+-N concentrations stabilised around 25 mg/L by week 12. Less than 10 mg/L $NO_3^- + NO_2^- - N$ occurred in columns amended with 15% and 30% fly ash, suggesting good denitrification being established. In these columns, some two thirds of the applied nitrogen was effectively being removed apparently through nitrification and denitrification. Nearly all NH_4^+-N removed in the unamended sand column also appeared as $NO_3^- + NO_2^- - N$. The results clearly indicate the beneficial effects of soil amendment by fly ash for effecting either complete nitrification or moderately high nitrogen removal through nitrification followed by denitrification, depending on the amount of fly ash present.

Other characteristics of treatment effectiveness

Effluent soluble COD in columns stabilised at around 40 – 60 mg/L by week 12 meaning that only a minor fraction of the applied soluble COD was being removed in all of the columns. The low biodegradability of the applied COD is the likely reason for this. The comparable removal efficiency achieved by all columns indicated that fly ash amendment did not greatly influence COD removal characteristics during soil infiltration.

With the 30% fly ash supplemented columns, progressive clogging of the column was observed to adversely affect the infiltration rate until after the eighth week, continuous ponding was noted on the surface. The clogging is attributed to low porosity of the medium and for this reason, fly ash amendment above about 15% is considered undesirable from an operational point of view.

Of the 10 metal ion concentrations monitored in the infiltration column effluents, only Ca and Se levels were substantially higher than in the septic tank effluent feed. Ca concentrations ranged up to 150 mg/L from columns with 5% fly ash amendment while Se concentrations were between 240 and 450 µg/L. The latter amounted to 5 to 10 times the permitted concentrations in drinking water (Lykins and Clark, 1994). The absolute levels are still considered quite low and if adequate dilution factor is available in the receiving water should not be a major environmental concern with the disposal of treated effluent.

CONCLUSIONS

The results of the foregoing investigations have clearly elucidated a high potential for beneficial utilisation of alkaline precipitator fly ash for improving effluent treatment efficiency, especially with respect to enhanced phosphorus and nitrogen removal. In particular, it has been shown that fly ash amendment of sandy soils with poor natural capacity for nutrient attenuation can result in significantly improved treatment performance during infiltration through amended filter beds. Sand amended with up to 15% fly ash appears to be suitable for making up improved infiltration media for septic effluent disposal. Long term performance characteristics of fly ash amended soils need to be evaluated in further studies before it can be recommended for critical applications requiring guaranteed performance.

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AEROBIC WASTEWATER TREATMENT SYSTEMS IN INDIGENOUS COMMUNITIES: APPLYING THE LESSONS FROM EXPERIENCE

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ABSTRACT

The Centre for Appropriate Technology Inc. (CAT) has been approached several times in recent years to comment on aerobic wastewater treatment systems (AWTS) proposed for use in Aboriginal and Torres Strait Islander communities. Since their introduction to Australia, AWTS have been able to produce high quality effluents under certain conditions, however, overall they have a chequered history in their applications with non-Indigenous communities. This paper presents a case study of AWTS installed in an Aboriginal community in Far North Queensland. Problems experienced are similar to those observed with AWTS in non-Indigenous communities, with additional complications. Many factors contribute to the poor performance of the AWTS in this community. The discussion here outlines the problems relating to AWTS in remote Aboriginal and Torres Strait Islander communities and presents some possible solutions. It concludes that AWTS are not an appropriate technology for Coen.

INTRODUCTION

This paper discusses Aerobic Wastewater Treatment Systems (AWTS) and their application in remote Indigenous communities. A case study is presented from Coen, a remote town located on Cape York Peninsula in far north Queensland. A brief outline of AWTS is provided followed by technology choice issues relating to Indigenous communities. The environmental and social conditions in Coen are discussed with a history of the AWTS installed in Coen. An analysis is undertaken of the reasons for these systems' poor performance over the past 4 years. The lessons available from the experiences in Coen are presented in the hope that they will be useful for future projects involving technology transfer of wastewater treatment systems to remote Indigenous communities. Recommendations are made for improving the AWTS performance in the short term. It seems likely that some form of off-site treatment and disposal of wastewater will be needed in the longer term, particularly during the wet months of the year.

BACKGROUND

In recent years the Centre for Appropriate Technology Inc. (CAT) has been requested by the Aboriginal and Torres Strait Islander Commission (ATSIC) to comment on the suitability of particular AWTS for use with remote Aboriginal and Torres Strait Islander communities. The Cairns office of CAT has assessed AWTS in a remote community on Cape York Peninsula in Far North Queensland. This assessment has raised concerns within CAT regarding AWTS in Indigenous communities. In addition, CAT has been involved in the research, design and implementation of other wastewater treatment systems during its years of working with remote Indigenous communities. There are invaluable lessons available from the 4-5 years of experience with AWTS in Coen. These lessons may be integrated into future technology delivery mechanisms so that the same mistakes are avoided with AWTS or any other wastewater treatment system in other locations.

More conventional treatment systems have a long history of development experience. Pit toilets have been widely used throughout Australia and the Developing World. Through many applications and years of trials, observations and experience this technology is now well understood. These systems can be installed taking account of social and cultural factors and their performance and outcomes can be reasonably predicted. Septic tanks have been used by non-Indigenous and Indigenous communities in Australia for over 50 years. They have been a useful wastewater treatment technology in the past but are now showing limitations in a new environmental management era. Unfortunately, the use of septic tanks with Indigenous communities has not been overly successful. It is only in recent times that active research of the background issues and performance of septic tanks with Indigenous communities is underway (Khalife, Dharmappa and Sivakumar, 1997).

It is with these experiences in mind that it seems necessary to be more rigorous in understanding the technology and the technology transfer process before attempting to implement a new technology into remote communities. It must be possible for an understanding of AWTS with respect to remote Indigenous communities to be developed in a much shorter period than has been necessary for pit toilets and septic tank systems. Technology transfer is a long and complicated process, however, this process can be enhanced by utilising lessons learnt from past experience. It is this experience which should inform decisions regarding the future use of these systems.

The objectives of this conference state that "it is .... imperative for us to develop a new generation of innovative environmental technologies to meet the challenge of small scale decentralised treatment". There is a perception that AWTS are an ideal small scale environmental technology for wastewater management. It is argued here that this may be the case in some circumstances, however, evidence is building that they are not an appropriate technology for remote Indigenous communities. There are lessons that have been learned about AWTS that should temper any ideas that *"a new generation of innovative environmental technologies"* is already available for remote Indigenous communities.

Aerobic Wastewater Treatment Systems (AWTS)

AWTS are a small package treatment plant with some models using activated sludge processes similar to large urban sewage treatment plants. Treatment is provided using a combination of pumps, fixed media, air blowers, fans and rotating discs. The treatment process involves primary and secondary treatment followed by chlorination and disposal by spray irrigation. AWTS are a relatively new form of technology for wastewater treatment. They have been around for 15 years or so and only in widespread use since the late 1980s. The concept originated overseas and has been developed into many different models here in Australia.

The aerobic environment of AWTS allows the treatment process to be much faster and more comprehensive compared to the anaerobic conditions in septic tanks. Consequently, greater levels of treatment can be achieved with shorter residence times and smaller capacities in the treatment units. The treatment, however, is generally limited to organic loads and pathogenic organisms (BOD_5, faecal coliforms, suspended solids) and has little effect on reducing nutrient levels.

The effluent quality produced from AWTS can be very good when the system's design conditions are maintained during use and appropriate and regular operation and maintenance needs are fulfilled (BOD_5=20mg/l, SS=30mg/l, faecal coliforms<30cfu/100ml). Geary and Gardner, (1996) and Petrozzi and Martens (1995) provide indicative figures of effluent qualities that are possible from these units.

The characteristics and requirements of AWTS include:
- units are sized for specific household loadings
- perform best with a reasonably consistent influent volume and quality
- need adequate primary treatment of influent prior to entering aerobic treatment chamber
- require a constant, reliable electrical power supply,
- need regular maintenance and servicing
- need a constant chlorine supply
- do not respond well to shock loads or periods of inactivity
- do not provide removal of nutrients from wastewater
- effluent is disposed of either by spray irrigation or subsurface disposal
- require adequate effluent disposal areas
- installation of units should involve education and training for residents and maintenance staff

The final treatment process after settling and clarification is chlorination. The effluent quality can be of a standard suitable to spray irrigate after the contact chlorination. However, most authorities require irrigation areas to be restricted to public access and irrigation methods and times designed to ensure there is no escape of irrigated effluent. If chlorination is not undertaken the effluent can be disposed of using subsurface irrigation or underground absorption trenches or beds.

There have been many problems with the recent widespread use of AWTS. The vast majority of installations have been to non-Indigenous households. The problems have included component failure, inadequate design and planning, lack of maintenance, inappropriate user behaviour and a lack of sufficient education and training for users, maintenance contractors and regulatory authorities. There has been a perception that AWTS can simply be installed and not thought about again. All of these problems have lead to poor overall system performance. (Hawkesbury-Nepean Catchment Management Trust, 1995, Langhorne et. al., 1995, Petrozzi and Martens 1995, Sawtell 1997, Schobben 1997)

These problems have produced a mixed response from regulatory agencies and local councils. In some areas new installations have been banned altogether, some councils have taken over the maintenance programmes while other councils have set up a system of registration for contractors servicing these units. In addition, some councils have established a system requiring householders to present regular maintenance certificates for their AWTS.

The manufacturers of AWTS have responded to these problems by developing guidelines for system design, operation and maintenance in an attempt to raise the standard of systems installed and the services provided by manufacturers and installation contractors. This approach seems to have improved the standard of AWTS available. The long-term future of AWTS is still unclear, however, the manufacturers efforts have been a good response to ensure their systems have a role in the area of on-site wastewater management.

Technology Choice and Transfer in Aboriginal and Torres Strait Islander Communities

Most technologies are designed and developed in a particular social and cultural context. There are many differences to be considered when introducing a technology into Aboriginal and Torres Strait Islander communities. Many technologies have developed around an understanding of non-Indigenous communities and they cannot be simply implemented directly into Indigenous communities.

Technology choice and technology transfer are processes that can produce the appropriate technology for a particular situation. Many factors need to be considered in selecting or developing a technology, including social, cultural, economic, political, technical and environmental issues. There is not a specific hardware or technical solution which should be used for every situation. Technology transfer is also a process with several stages. Pre-transfer activities, tasks during implementation and post-implementation actions are all necessary aspects. Research and data collection, resident involvement, education and training and operation and maintenance support are all aspects of technology transfer (Fuller, 1997).

In the case of wastewater treatment systems, existing standards, guidelines and technical practices need to be interpreted to reflect the differences between non-Indigenous and Indigenous lifestyles and conditions. In the Northern Territory, for example, Territory Health Services has developed its own code of practice for wastewater management based on existing Australian standards and other guidelines which reflect the conditions and realities in remote communities. It has used data relating to the social, cultural and environmental circumstances of its Indigenous communities. For example, septic tanks sizes are larger, all-waste tanks are not permitted, the systems must be split blackwater and greywater systems and the inflow allowance is double for Indigenous households compared to urban households (Territory Health Services, 1996).

There are obviously many aspects in the social, cultural, political and environmental arenas which are unique to Indigenous communities. There are some factors which can be identified that will affect the process of technology choice and the appropriateness of particular technologies for wastewater management. A brief list of such factors may include:

- frequent high household numbers and sizeable variations over time,
- low home ownership levels, families commonly living in State government housing, a high level of housing shortage,
- high housing densities in State government housing developments which are associated with small block sizes,
- high mobility between households and communities,
- limited financial resources and capacity to pay for services including rent, electricity, water, rates and maintenance costs,
- possibly low levels of public health awareness,
- limited employment opportunities,
- large, young families,
- limited education and experience in technical areas,
- remote locations,
- poor quality power supplies, and
- harsh environmental conditions.

The Characteristics of Coen

Outline

Coen is a small town located on Cape York Peninsula in far north Queensland. It is approximately 500 km from Cairns and is situated on the Peninsula Development Road. Water is reticulated throughout Coen by Cook Shire Council (CSC). Electrical power is generated and reticulated throughout the town by Far North Queensland Electricity Board (FNQEB) and is provided at the State Equalised Tariff rate. All household wastewater is treated and disposed of on-site. Some households have septic tanks and disposal trenches only and in addition there are approximately 25 AWTS in use.

The health clinic staff in Coen have been concerned for some time at the prevalence of gastro-intestinal diseases related to domestic wastewater particularly in children. The vast majority of these presentations to the clinic correspond very strongly to the wet season. Many parents acknowledge that their children often play in or near the disposal trenches or the AWTS. Advice from health clinic staff to parents to stop their children playing in such areas has been reported to have reduced the incidence of sickness.

Details of the Town

Coen is regularly isolated by road for 3-4 months of the year during the wet season. The Peninsula Development Road becomes impassable both to the south and the north. An air service to Cairns continues year round as there is a sealed airstrip at Coen.

Approximately half the area of Coen township is reasonably flat or gently sloping. The remaining sites are quite steeply sloping blocks. There are several creeks and gullies that run through or around Coen which flow quite strongly during the wet season. There is concern over levels of effluent pollution in these waterways as many people, especially children, swim in the creeks and gullies. The soil around Coen is highly impermeable and contains significant levels of clay with bedrock also close to the surface. In one small area there is reasonably permeable soil to a depth of approximately 1.0 m which is underlain by clay.

Coen experiences a very high, intense rainfall for a short period of approximately 3 - 4 months during the wet season. The watertable rises considerably during the wet season and lies very close to the ground surface. The soils become saturated and some tend to anaerobic conditions particularly areas in the vicinity of disposal trenches. During the dry season evaporation, transpiration and soil absorption ensure there is virtually no sign of wastewater at the ground surface.

Many houses are located on sloping blocks with inadequate road kerbing and stormwater drainage facilities. Sites adjacent to gullies have slightly better drainage but soils remain quite wet during the high rainfall period. Many houses are also poorly sited on the blocks and blocks are very small, usually 20 m x 40 m. The topography, house position and block size often make it impossible to suitably position adequate disposal trenches within block boundaries. Attempts have been made but this has often meant trenches are not appropriately sized and graded due to the space constraints.

There is very limited roadwork and associated infrastructure around the town. Several streets with State government housing are unsealed and poorly graded. During the wet season many of the roads have constant standing water ponding along their edges which may be contaminated with partially treated or untreated effluent. In addition, untreated or semi-treated wastewater has been observed surcharging from trenches and draining along street edges.

Due to its remote location and problems with access, there are difficulties in obtaining tradespeople to service Coen on a regular basis. It is possible to get tradespeople for large contract works but not for intermittent works such as servicing infrastructure. This shortage of skills has implications for the quality of work undertaken and the on-going operation and maintenance costs associated with ensuring adequate performance of infrastructure.

Coen is not an Aboriginal community although the majority of the population are Indigenous people. It has a total population of around 320 people. The few non-Indigenous residents living in and around town are either service providers including shopkeepers, health workers, accommodation providers and policepeople or farmers on surrounding properties.

The housing in Coen is a combination of rental houses owned by the State government and Coen Regional Aboriginal Corporation (CRAC) and privately owned houses. The vast majority of housing is State government owned by the Department of Public Works and Housing (DPW&H). People pay rent and the State government undertakes the maintenance and housing management. There remains a constant housing shortage in Coen, consequently, household numbers can be very high particularly during the wet season. These household numbers are also subject to large variations over time. Many residents are associated with farms or land in the surrounding area to Coen and often spend extended periods of time away from town. Most people return to Coen during the wet season when access to their land is restricted.

Income levels are generally very low in Coen. Most people are employed on the ATSIC sponsored Community Development Employment Programme or are on pensions. These low incomes combined with large families, high household numbers, poor health and a remote location create problems for covering costs such as food, rent, transport and services. It is not uncommon for households to have the electricity disconnected due to an unpaid bill. The inconvenience of disconnection is compounded with an additional charge for power reconnection.

Aerobic Wastewater Treatment Systems in Coen

There are approximately 25 State government houses in Coen. These houses were originally serviced by septic tanks and disposal trenches for blackwater only. Greywater was disposed of informally on-site. Despite only handling the blackwater there had been problems with the septic tank and trench performance. Four years ago AWTS were installed to each house. At this stage all internal greywater drainage was plumbed to the existing septic tank. Since installation there have been constant problems with the performance of the AWTS and the disposal trenches. Septic tank and disposal trench systems are also performing poorly.

The AWTS in place in Coen consist of the existing septic tank, which acts as the primary treatment chamber, an AWTS which undertakes secondary treatment, settling, clarification and chlorination and a holding tank from which effluent is periodically pumped to disposal trenches. Spray irrigation was not recommended due to a lack of sufficient space on each block. The final effluent is disposed of underground and consequently a decision was made not to chlorinate the effluent prior to pumping to the trenches. However, once it was observed that the trenches were surcharging and effluent was rising to the surface chlorination of the effluent was recommenced.

It was reported that, due to the very dry 8 months of the year and regular imposition of water restrictions, the residents of Coen were in favour of AWTS because of their potential to produce an effluent that could be re-used for irrigation purposes. This is suggested as one of the reasons AWTS were installed in Coen. However, when the AWTS were installed there does not appear to have been adequate training and education provided to the residents of the houses or to housing maintenance staff.

It is not uncommon for houses in Coen to have the electricity disconnected. When this occurs the AWTS has no separate electrical power supply. Without electricity the aerobic treatment processes stop and pumping from the holding tank to the disposal trenches ceases. The untreated wastewater backs up inside the unit until it seeps from the top and sides. A similar situation occurs when either of the two internal pumps fail. Pump failures have been very common in Coen. The supplier of the units has commented that pump failure rates have been extraordinarily high in Coen compared to units installed around Cairns. This has raised questions over the quality of the electrical power supply in Coen. A shortage of qualified tradespeople to service the units, difficulty in obtaining parts and replacement pumps and the remoteness of Coen have all hindered the process of keeping the internal pumps in working order. Both the electricity disconnections and pump failures have contributed significantly to the overall poor performance of the AWTS.

With the high watertable conditions in the wet season any form of subsurface effluent disposal will be problematic for those months. As mentioned earlier, the health clinic records document sickness presentations corresponding directly with the wet months of the year. During the drier months the disposal trenches appear to be functioning reasonably well and minimising human contact with the effluent. However, the AWTS performance and reliability is not related to the wet season except for the additional loading during this period. It is the failure of the units together with the failure of the trenches which leads to the pronounced increase in sickness levels. If the units were performing adequately and only the trenches were failing during the wet season the consequences of trench failure would not be as severe. The units fail year round and the trenches fail seasonally. For the period when the trenches are seemingly performing adequately the units' poor performance is not so dramatically reflected in presentations to the clinic. The trench failures highlight and focus attention on the failure of the overall wastewater treatment system.

DISCUSSION

The failure of AWTS in Coen is related to many factors some of which are technical and some are social and behavioural. The units have failed primarily because of:

- fluctuations of influent quantity and quality
- overloading of system
- poor septic tank performance
- irregular electricity supply
- regular internal pump failures
- inadequate disposal trench design and construction
- poor environmental conditions for on-site wastewater disposal
- very small block size leading to insufficient disposal areas
- inappropriate resident behaviour
- irregular servicing
- shortage of skilled tradespeople to undertake repairs and maintenance when needed
- lack of sufficient resident and maintenance staff education and training

No one reason is particularly more significant with many combining to generate a poor situation. When considering an appropriate wastewater system for the households in Coen it is essential to consider background issues such as those outlined earlier. They should be used to develop a picture of the context within which the systems will need to operate. The lifestyles and behaviours in many houses in Coen are quite different to the patterns in a suburban or rural non-Indigenous household.

The high resident numbers and fluctuations cause wastewater volumes to vary significantly both within a household and between households. Houses are sometimes vacant for extended periods which means the aerobic micro-organisms (activated sludge or biofilm) in the units reduce in number and activity. When the house is occupied again there is insufficient sludge to treat the incoming organic loads. Over time the sludge quantity increases and will be maintained if the influent stream remains constant. A similar pattern occurs with electricity disconnections and reconnections. During the peaks in occupancy levels the influent loadings not only overload the activated sludge quantities present but the volumes are often in excess of the capacities of the septic tanks and AWTS. These patterns can repeat throughout the year and are independent of the other environmental factors causing system failure.

As discussed earlier, household behaviour is often different in Indigenous households compared to non-Indigenous households. In addition, access to household infrastructure is quite different in Indigenous communities and this often has an effect in areas such as wastewater volumes generated. Therefore, a system designed for non-Indigenous households may not suit an Indigenous household which is comparable in size and location. For example, people from the extended family may all do their clothes washing in one house because it is the only house with a working machine. Recent research has shown that washing machines in Aboriginal households are often used up to seven times more than the non-Indigenous Australian average (Lloyd, 1997).

The existing septic tanks have been utilised as part of the treatment system. These tanks were installed to treat blackwater only and to suit an average 2-3 bedroom household. They certainly were not sized for the household numbers that periodically occur in Coen. Consequently, their capacity is insufficient to provide the level of primary treatment needed before the wastewater enters the AWTS. In addition, the septic tanks are not desludged regularly which further reduces their capacity. This limited septic tank capacity has caused greater levels of solids contamination to enter the AWTS than their design levels. Solids have built up at a greater rate in the AWTS requiring greater levels of treatment. The units require desludging more regularly to keep solids levels down and allow the aerobic sludge to treat the wastewater to design standards. The AWTS are designed for approximately 10 residents, however, household numbers often exceed 10 residents which means the AWTS are too small for these peaks in household populations.

The current Australian Standard (AS1547) for disposal of effluent from AWTS requires an irrigation area of 200 m^2. This area should be remote from commonly used yard areas and have some sort of barrier to restrict easy access. The blocks in Coen are generally 20 m x 40 m in size and consequently there just is not the space for such an arrangement on each house block. Rather than irrigate, it was decided to install disposal trenches in order to save space, save on chlorination costs and keep the effluent below the surface to avoid human contact. However, with the soil conditions and effluent volumes present in Coen disposal areas need to be significantly larger than is possible on an 800 m^2 block. On-site disposal of effluent under these circumstances is simply not possible on an 800 m^2 block, unless effluent is treated to near potable standard for re-use both inside and outside the house (Geary, 1992).

There are alarms fitted to the AWTS to signal when any component failures occur. The remoteness of Coen and the shortage of available tradespeople means units cannot be repaired when the alarms sound. Even with regular servicing components fail between services and there is no one available to fix or replace them. Residents have become very unresponsive to the alarms because their experience is that units will not get attended to when they do report problems. Access to qualified people to repair and service the units is a major component in ensuring the AWTS function consistently and reliably. Initially in Coen the servicing was not undertaken as specified, however, this has since improved. The maintenance staff report to undertaking the required servicing works, however, they comment that it is between the services when problems occur. They say they would have to be paid to be available at all times if they are to keep the units consistently operational. The remote location means it also takes time to get the replacement parts from Cairns or Brisbane to Coen. In addition, maintenance work to other infrastructure is often not completed. For example, repair of leaking taps, desludging of septic tanks or renovating of disposal trenches are all activities which would improve the performance of the overall system. Clearly , remoteness is a design constraint that requires a great deal of thought before commissioning a new technology. The relative value of innovation is dependent on location.

During the wet months there is a high likelihood of trench failure. The risks from this failure need to be considered. There should be a strategy for dealing with this likely seasonal failure. One approach may be to spray irrigate on vacant land or the racetrack for the wet months of the year and revert to on-site trench disposal once the ground has dried out. Even with this approach some rectification work to trenches would be needed.

It has been suggested that there was insufficient user and resident training and educational activities when the AWTS were installed. The old septics were plumbed to the new AWTS and new trenches were installed for each house. However, there was little effort invested in involving the residents in the process or explaining the new system and its requirements. A small amount of printed literature was available but no effort was made to ensure householders understood the new system and the requirements of living with an AWTS.

The high level of component failures that have been observed in Coen have also been reported from other remote locations in Australia where AWTS have been installed. One reason suggested for this is that systems and their components have been designed for the southern, east coast locations of Australia. Conditions in the tropical areas and remote desert sites in central Australia are very different to SE Australia. Factors such as temperature, humidity and power quality could all have an impact on the operation and life of AWTS and their components.

Interim Solutions in Coen

The following recommendations are given as possible solutions in the short term. The temporary nature of the interventions relates to the underlying fact that some form of off-site treatment and disposal will probably be necessary in Coen in the long term. Due to a significant lack of funds this will obviously be well into the future which means solutions are needed now for the AWTS. The future arrangements may or may not utilise existing infrastructure and the additions made in the short term.

The first step to improve wastewater conditions in Coen should involve some type of household survey to get a better understanding of population figures and fluctuations in the houses. This would provide useful data on the design conditions for the wastewater systems. It may show that only some houses have large peak populations and some have a relatively stable number of residents. This information could be utilised to target works strategically where they would have the greatest impact. This exercise would also raise awareness amongst residents to the issues of wastewater treatment and allow for some level of education and training with householders and maintenance staff.

The State government is considering installing separate electricity connections to the AWTS and paying these power bills. This should ensure the units have a consistent electricity supply. Comments by the manufacturer about component failure being significantly higher than in other locations raises questions over the quality of the electricity supply. This issue is still to be resolved and needs investigating thoroughly to identify whether it is a factor in component failure. These investigations should produce a strategy to minimise any electrical power quality problems for the AWTS.

It would be useful to either install larger septic tanks or a second tank in combination with the existing tank to provide more residence time prior to effluent entering the AWTS. This would aid in reducing shock loads and reduce solids loads to the AWTS. In addition, the AWTS need to be increased in capacity in order to manage the peak loading conditions. Access to parts also needs to be improved. The DPW&H and CRAC should consider purchasing common components that fail regularly to have easily available in Coen.

Another suggested strategy is to move the alarm and light indicators from the side of the units and install them inside the houses. This will certainly make them more noticeable, however, the issue is whether this would make people more likely to report a unit failure. As mentioned above residents are very unresponsive to the alarms. If the alarms are moved this needs to be accompanied with other strategies to ensure resident reports will generate a response. This may encourage residents to report failures.

Once the actual treatment side of the process is improved, it would be necessary to assess and remediate trenches. It is anticipated that significantly greater lengths of trenches will be needed with better positioning around the block. It is, however, highly likely that many sites will not be able to adequately fit the necessary additional trenches within the block boundary. It may be necessary to look at the possibility of irrigating good quality effluent onto surrounding land. This may become a strategy to be used only during periods of high rainfall and saturated soils.

From the management side there are several actions which would help to improve the situation. In terms of minimising overloading both the AWTS and disposal areas it would be useful to employ a plumber to check and fix all taps, showers, toilet cisterns and other fixtures prior to the start of the wet season. The work could be a joint effort by CRAC and DPW&H who are the major housing owners in Coen. The process should involve a local Coen resident to work with the tradesperson and learn the tasks to be undertaken. This person would be available in Coen when necessary throughout the year to continue with regular maintenance works. In addition, it may be necessary to arrange for a contractor to be available in Coen at short notice. For example, the CSC water supply technician may be able to do minimal works in emergency situations to ensure systems are kept in operational condition. In addition to these measures it is absolutely necessary to establish a contract with maintenance personnel to ensure regular maintenance is undertaken on all units.

Lessons Learned from Coen

The experiences in Coen with AWTS provide vital lessons that can inform future processes of technology choice for wastewater treatment systems in remote Indigenous communities. There are implications for the research and design, implementation and on-going operation and maintenance phases.

During the research and design phase the following aspects should be considered:
- know the recipient population well in terms of demographics and mobility patterns,
- interpret design standards and guidelines to suit the location and community,
- undertake necessary data collection where it does not already exist,
- the design parameters for the entire year should be well understood (e.g. climatic conditions, user patterns etc)
- understand the existing infrastructure and any limitations that may exist,
- understand the O&M needs and design programmes to ensure they can be met either by a local person or contractor,
- identify the feasibility of a maintenance contract, both cost/affordability and physical accessibility,
- research the availability of spare parts and staff to install on an as-needed basis not only during the regular servicing visits,
- a process of checking the suitability of a proposed technology against the location,
- a process to assess the suitability of the proposed technology on a social, cultural and environmental basis, and
- strategies for the implementation phase to enhance technology transfer processes.

There are certain things that can be done during the implementation phase that will enhance the sustainability of the project and the infrastructure installed. Involving local people wherever possible is essential. Ensuring adequate education and training activities are undertaken as part of the implementation works will also increase success. In addition, appropriate attention should be given to technology transfer activities identified during the design phase.

Operation and maintenance (O&M) needs should be designed to suit the local conditions. The range of issues presented in this paper provide an indication of the considerations that should inform the design of any O&M methods and delivery structures. This phase is essential for the long term performance of any technology. Considerable planning and preparation should be devoted to the design of these aspects of the work.

The successful installation of AWTS is dependent on a thorough technology transfer process. This process requires additional understanding when an Indigenous household is involved. Many technologies fail because of insufficient attention to the recipient population and how well the technology matches their situation. The failure of many AWTS could be attributed to poor technology transfer processes both with non-Indigenous and Indigenous households.

CONCLUSION

The AWTS technology may be suitable for particular applications and situations. However, not all technologies are suitable for all situations. The factors discussed above should be considered when choosing, designing and/or transferring a technology to an Indigenous recipient group. The process of technology choice is about developing and identifying the most appropriate technology for a particular location. Appropriate technologies are not a group of hardwares to be selected and applied in different locations. Appropriate technologies are those selected on the basis of a rigorous process of evaluating many factors including social, cultural, political, economic, technical and environmental aspects in a particular location. Many of these issues were not considered when the AWTS were installed in Coen and the technology is deemed inappropriate in that community at this point in time.

It is important to remember that the technology transfer process is as important as the technology itself. The significant problems with AWTS throughout Australia, in both Indigenous and non-Indigenous communities, can in part be attributed to a poor technology transfer process.

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SLUDGE TREATMENT AND NUTRIENT REMOVAL

PHOSPHORUS REMOVAL FROM THE EFFLUENT OF A BREWERY WASTEWATER TREATMENT PLANT

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ABSTRACT

Brewery wastewater final effluent treated by activated sludge, although adequately reducing BOD_5 and suspended solids, contains significantly high levels of phosphorus (1 500-2 000 µg/L). Further phosphorus removal was achieved by additional chemical precipitation and/or soil amendment.

Red Mud neutralised with Gypsum (RMG) and red sand were found to be effective soil amendment media. The adsorption of approximately 0.076 mgP/g of red sand was achieved before media loss of phosphorus adsorption capacity, determined by phosphorus breakthrough in the effluent. The addition of 10% RMG was found to increase the adsorption capacity to approximately 0.114 mgP/g of the mixed media. A further increase of RMG to 20% and 30% slightly increased phosphorus adsorption capacity, however this significantly decreased the flow rate through the media.

With chemical precipitation, up to 90% phosphorus removal was achieved using 3 moles of aluminium sulphate or 6 moles of ferric chloride per mole of phosphorus.

To ensure phosphorus discharge levels meet regulatory requirements, combined chemical precipitation followed by soil amendment may be appropriate. Soil amendment will reduce the risk of phosphorus level fluctuation which occurs with chemical precipitation alone. In turn, chemical precipitation prior to soil amendment will significantly decrease phosphorus and suspended solids levels in the effluent. The infiltration rate and life span of soil media will thus be prolonged. The need to replace the media can be significantly reduced (or eliminated) if appropriate grass sp, which can remove 40-60 kgP/ha/yr, are planted in the disposal area.

INTRODUCTION

Due to increasing concern regarding the deteriorating condition of the Swan-Canning River system, Western Australia, due to nutrient loads, Swan Brewery at Canning Vale (Western Australia) was targeted as a significant point source. Although the wastewater from the brewery undergoes treatment using the activated sludge process, which adequately reduces BOD_5 and suspended solids, the final effluent was found to contain levels of phosphorus significantly higher than that allowed by the relevant regulatory authorities.

Phosphorus can be removed from wastewater by chemical precipitation with aluminium, iron, or calcium salts (Tebbut, 1991). To achieve an adequate and economic reduction in phosphorus level is dependent upon various factors primarily pH, suspended solids concentration and other chemical constituents of the liquor.

Alternatively, phosphorus may be removed by passing the wastewater through a bed of phosphorus adsorbing media. The use of RMG and red sand as soil amendment media for removal of phosphorus is well established (Cheung & Venkitachalam, 1994; Ho *et al.*, 1992; Ho & Mathew, 1994; Sampson, 1994). The adsorption and infiltration capacities are, however, dependent upon the characteristics of the treated wastewater (e.g. suspended solids, pH, etc).

Thus this study aimed to investigate and evaluate the most effective and economic conditions for further removal of phosphorus from the brewery final effluent by chemical precipitation and/or soil amendment.

EXPERIMENTAL PROGRAM

Phosphorus Removal by Soil Amendment

Column test experiments were carried out using perspex columns of 2.5 cm internal diameter. Five trials of different mixed media were used:

- Trial A: 100% Red sand
- Trial B: 10% RMG and 90% Red sand
- Trial C: 20% RMG and 80% Red sand
- Trial D: 30% RMG and 70% Red sand
- Trial E: 30% RMG and 70% Bassendean sand (find silica sand extensively leach with negligible ion exchange or adsorption capacity)

These soil mixtures were packed to 10 cm depth and had a bulk density of 1.5 kg/m^3. To ensure this, the columns were packed with 1 cm or 2 cm increments of the soil wetted with distilled water and compacting it to the desired depth increment. Three identical columns were used in each trial.

The water samples used in the experiment were taken from the final effluent of Swan Brewery wastewater treatment plant. The solution was fed to the column using a Mariotte bottle principle or syphon arrangement and water level of each column was maintained at approximately 20 cm above soil level.

The effluent from each column was measured for the rate of infiltration. Regular samples were analysed for orthophosphorus (a preliminary experiment showed no significant different between total phosphorus and orthophosphorus in the sample).

Phosphorus Removal by Chemical Precipitation

The chemical precipitation experiments, using aluminium sulphate and ferric chloride, used 500 ml samples which were subjected to a standard jar test (10 seconds of rapid mixing followed by 5 minutes slow mixing, then settling for 1 hour). Five dosages of each chemical flocculant were chosen to cover the optimum dosage range (1.5-3.0 moles of aluminium per mole of phosphorus, 2.0-5.0 moles of iron per mole of phosphorus; Eckenfelder, 1980). All experiments were carried out in duplicate. Average values are presented in this report.

RESULTS AND DISCUSSION

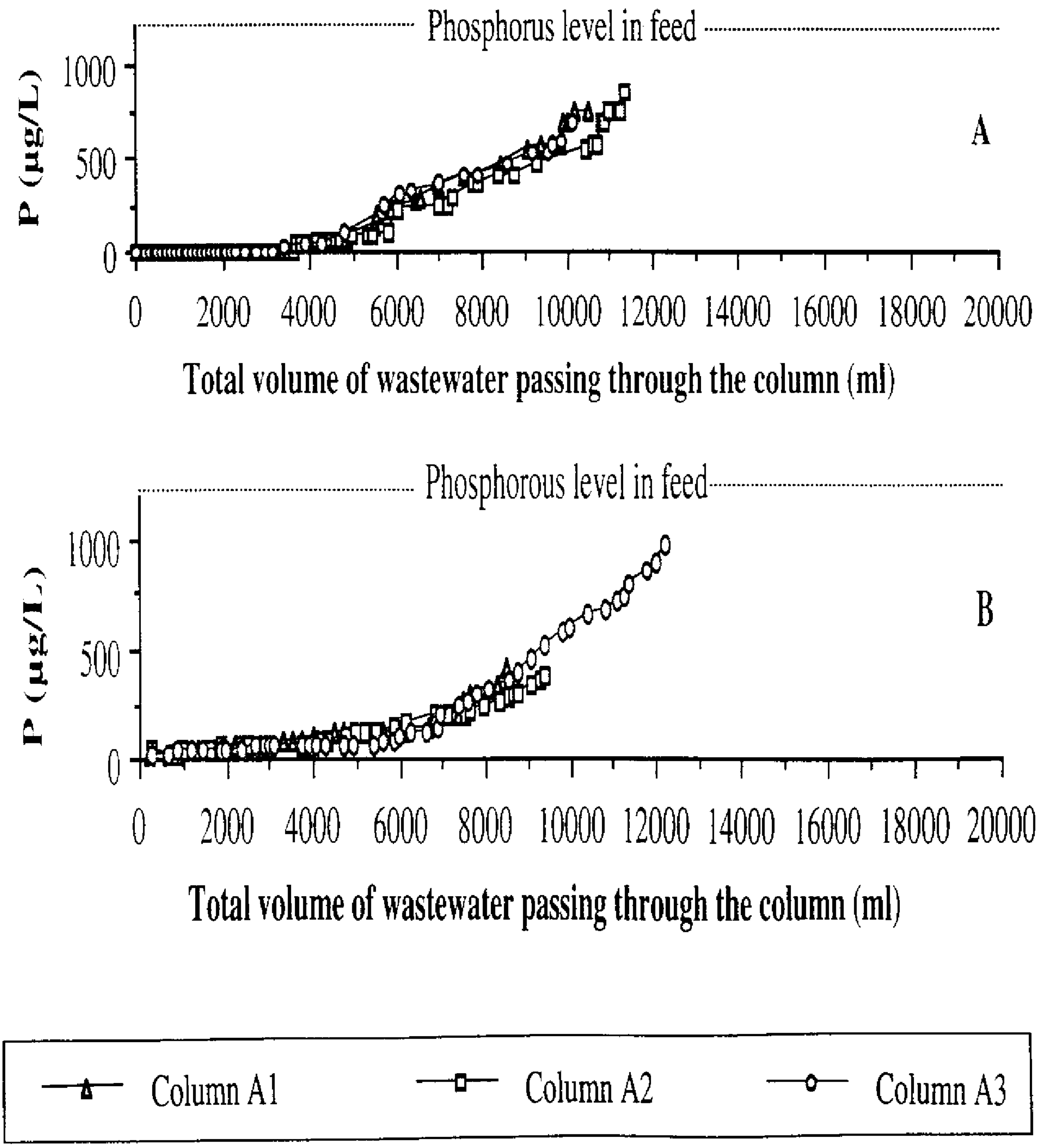
Phosphorus Removal by Soil Amendment

Table 1 shows characteristics of the samples, taken from the final effluent of activated sludge system. Phosphorus levels of the samples vary between 1200-1580 μg/L. Suspended solids were approximately 51 mg/L and pH was between 8.2-8.7.

Table 1. Characteristics of wastewater used.

Phosphorus (μg/L)	1200 - 1580
Suspended solids (mg/L)	42 - 60
pH	8.2 - 8.7

Figures 1 and 2 present the adsorption capacity and liquid flowrate for the five different packing media. As can be seen, the phosphorus adsorption capacity of all types of mixed media is considerable. Up to 99% phosphorus removal was observed initially, then phosphorus in the column effluent gradually increased depending on the type of packing media. Flowrate of wastewater through the media dramatically reduced with the addition of RMG.



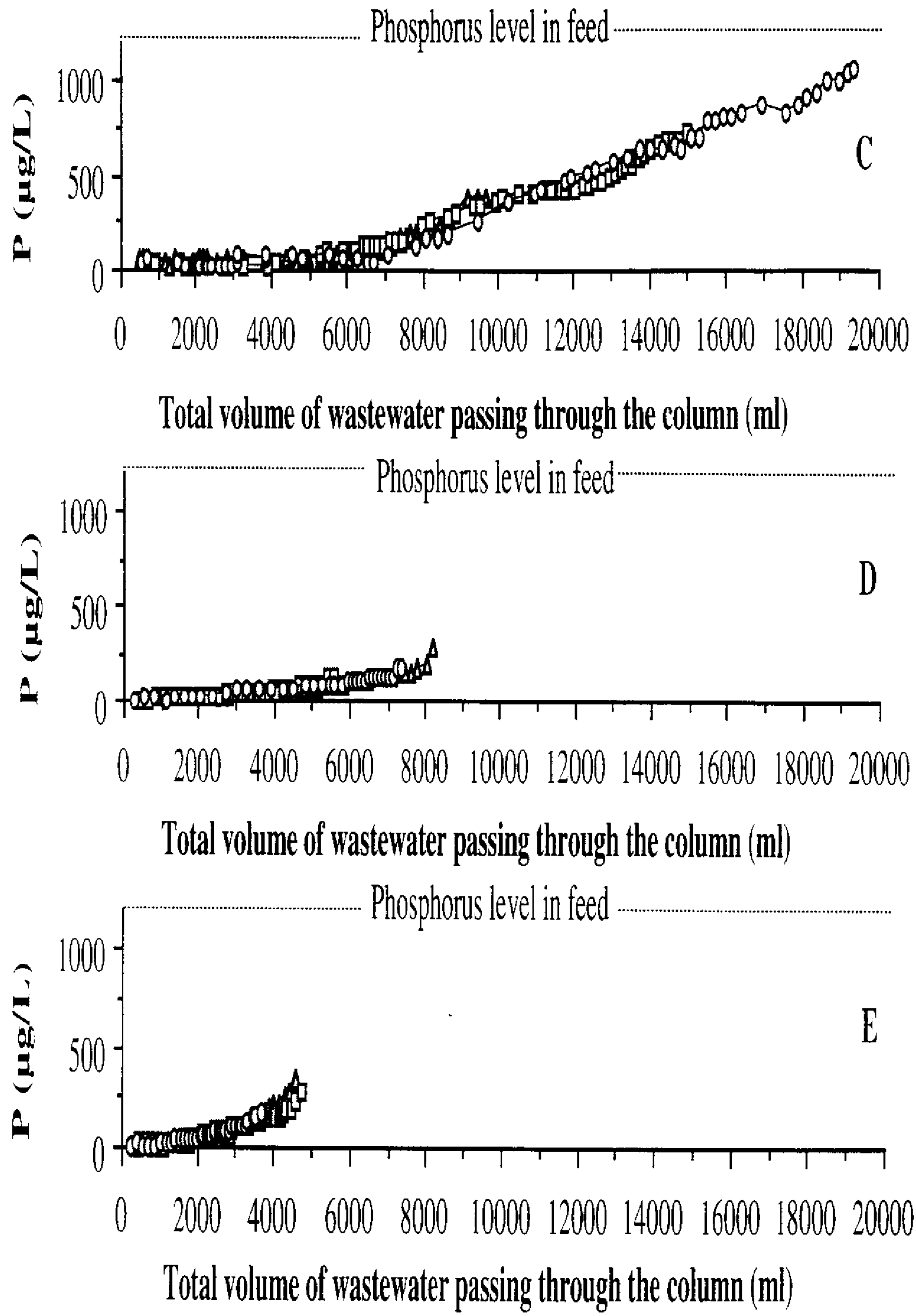
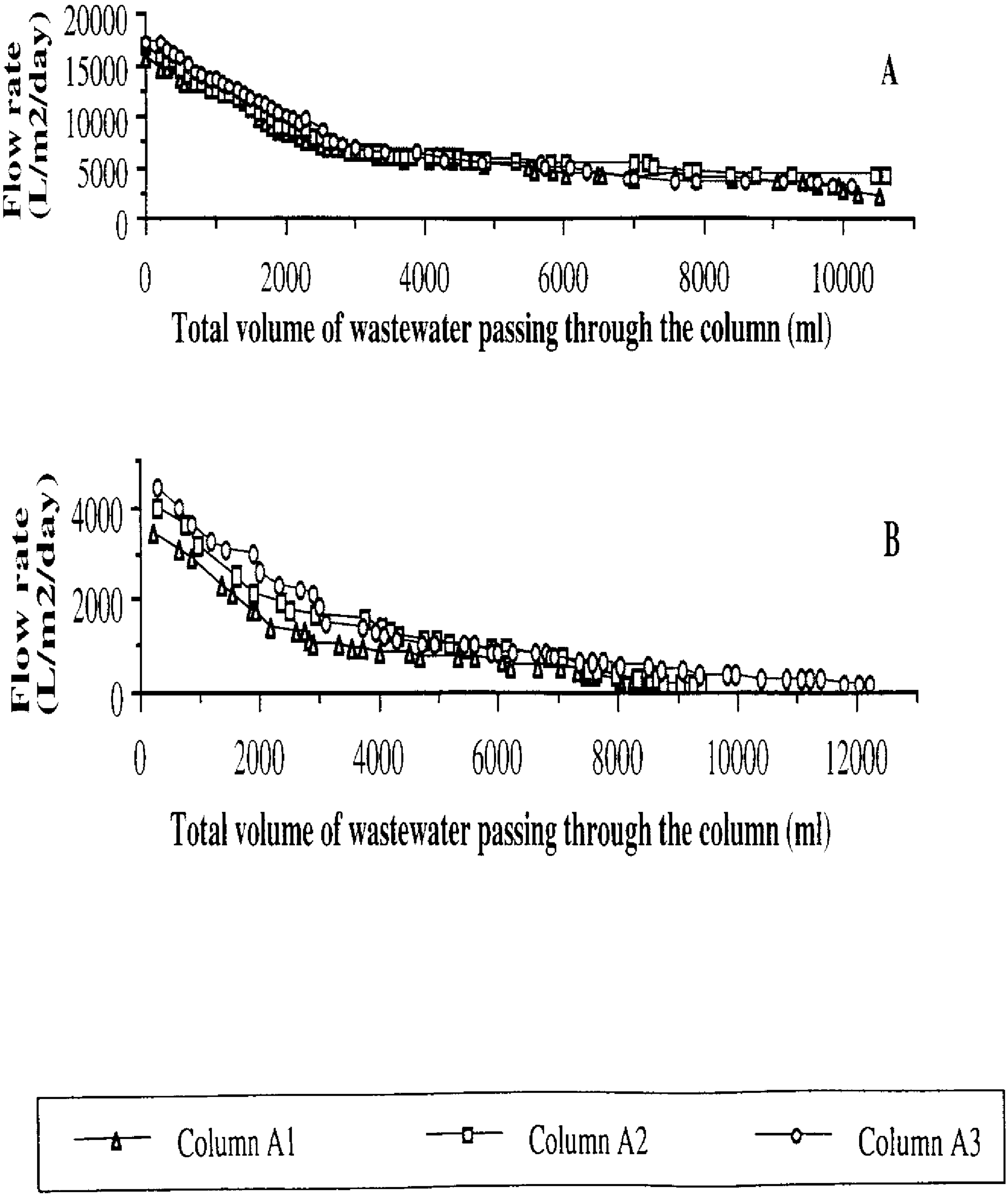


Figure 1 Phosphorus levels in the column effluent as a function of the cumulative wastewater volume through the column: Trial **A:** Red sand 100%; **B:** 10% RMG and 90% Red sand; **C:** 20% RMG and 80% Red sand; **D:** 30% RMG and 70% Red sand; **E:** 30% RMG and 70% Bassendean sand


Flow rate (L/m2/day)
20000
15000
10000
5000
0
A
0
2000
4000
6000
8000
10000
Total volume of wastewater passing through the column (ml)
Flow rate (L/m2/day)
4000
2000
0
B
0
2000
4000
6000
8000
10000
12000
Total volume of wastewater passing through the column (ml)
Column A1
Column A2
Column A3

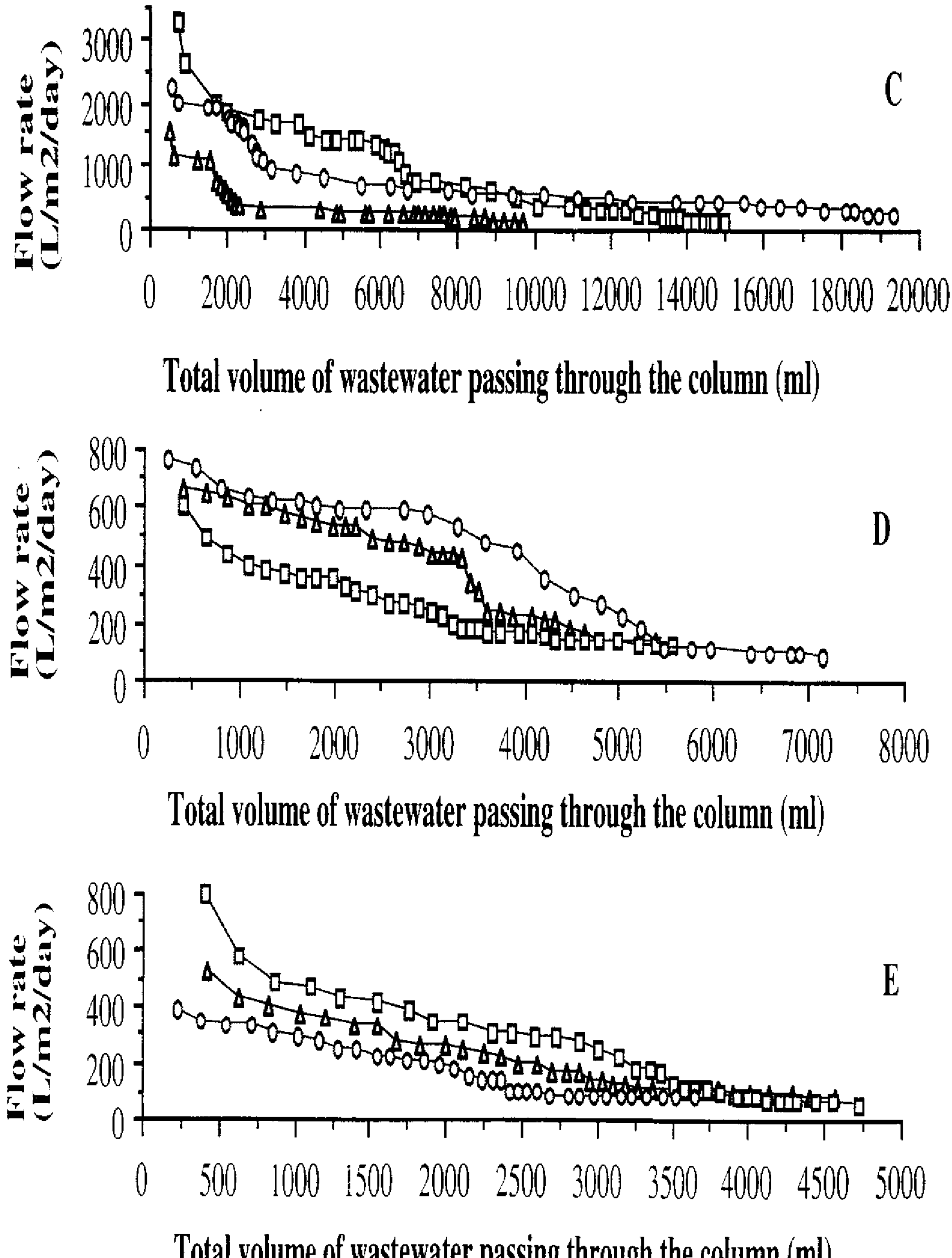


Figure 2 Flowrate of wastewater through the media as a function of the total volume passing through the column: Trial **A:** Red sand 100%; **B:** 10% RMG and 90% Red sand; **C:** 20% RMG and 80% Red sand; **D:** 30% RMG and 70% Red sand; **E:** 30% RMG and 70% Bassendean sand

Figure 1A shows the adsorption capacity of the red sand. Phosphorus breakthrough was observed to occur after a total volume of approximately 4 000 ml of sample had passed through the column. This is equivalent to the adsorption of approximately 0.076 mgP/g of red sand before media loss of phosphorus adsorption capacity, as shown by the gradually increasing phosphorus level in the effluent. Note that the flow rate of wastewater through red sand was approximately ten-times higher than that through a mixture of red sand (90%) and RMG (10%).

The addition of 10% RMG was found to increase the adsorption capacity of the media. Figure 1B shows the increased total wastewater passing through the media before phosphorus breakthrough occurred (shown as the increasing of phosphorus in the column effluent, indicating the media lose their adsorption capacity). The breakthrough varies from 4 000 ml with red sand alone to 6 000 ml with the mixture of red sand and RMG. As a result, the adsorption capacity increased to approximately 0.114 mgP/g of the mixed media.

Further increases of RMG first to 20% and then to 30%, slightly increased phosphorus adsorption capacity, as shown in Figure 1C and 1D. However, when RMG was increased to 30% the flow rate through the column slowed dramatically. This trial was therefore stopped when it reached an impractical flow rate of below 100 L/m^2/day (Figure 2D).

With the trial using mixed media involving Bassendean sand, phosphorus adsorption capacity was much lower than that involving red sand (Figure 1E). The infiltration rate was extremely low (Figure 2E), this trial was therefore terminated as a result.

Phosphorus Removal by Chemical Precipitation

The water samples used in this experiment contained 1.65 mg/L of phosphorus, 53 mg/L suspended solids, at pH 8.52.

Precipitation by aluminium sulphate ($Al_2(SO_4)_3.16H_2O$)

Figure 3 below showed the phosphorus removal by precipitation using aluminium sulphate at different pH. At the optimum pH of 6, a significant amount of phosphorus could be removed when the aluminium sulphate dose was increased to 100 mg/L (about 3 moles of aluminium per mole of phosphorus). Further increasing of the aluminium sulphate dose to 200 mg/L did not achive significantly further reduction in the phosphorus level.

By using sample without adjusting pH, or with pH lower than 6, the precipitation capacity was significantly reduced. This confirms the necessity of pH control during precipitation process.

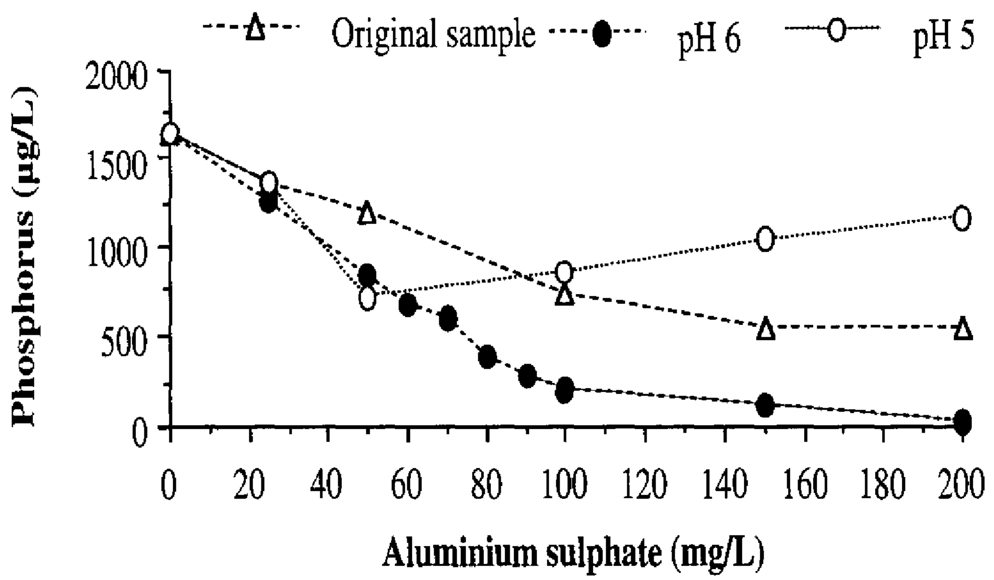


Figure 3 Phosphorus concentration in samples after precipitation

by aluminium sulphate at different pH values

Precipitation by ferric chloride ($FeCl_3$)

With ferric chloride, the optimum pH for phosphorus precipitation was found to be 5. Figure 4 shows a higher level of precipitation at this pH. The optimum dose of ferric chloride for this particular sample was 50 mg/L (approximately 6 moles of iron (Fe^{3+}) per mole of phosphorus). As with the aluminium sulphate, precipitation away from the optimum pH gave a significant reduction in phosphorus removal.

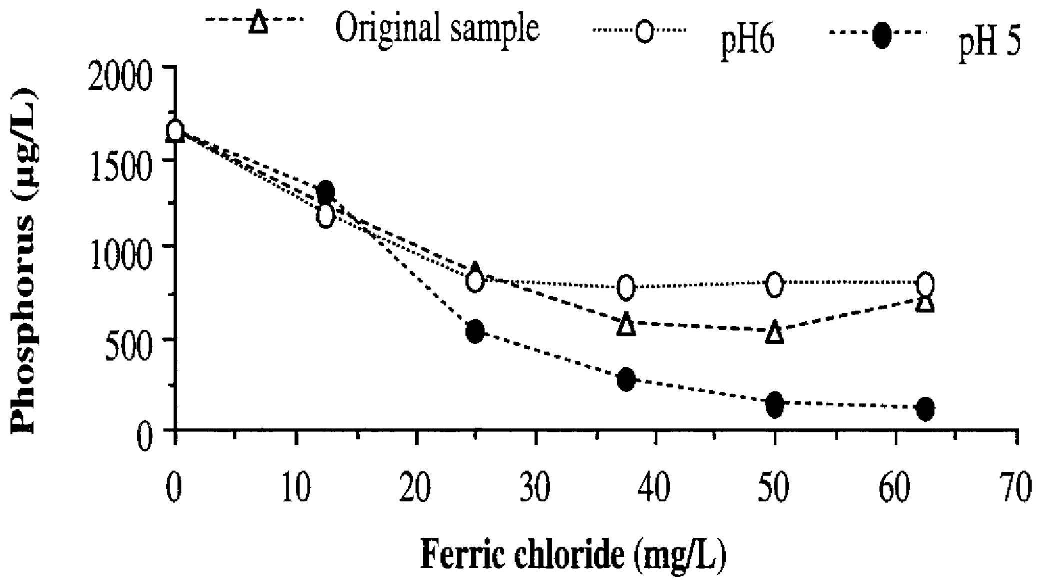


Figure 4 Phosphorus concentration in the sample after precipitated by ferric chloride at different pH

It should be noted that the chemical dosage used in this experiment (3 moles of aluminium sulphate and 6 moles of ferric chloride per mole of phosphorus) was slightly higher than the theoretical value (1.5-3.0 moles of aluminium sulphate and 2.0-5.0 moles of ferric chloride per mole of phosphorus; Eckenfelder, 1980). This is probably due to colour and suspended solids in the sample which consume some flocculant during the precipitation process (Eckenfelder, 1989).

GENERAL DISCUSSION

The results above show that phosphorus can be effectively removed either by soil amendment (passage through a soil bed) or chemical precipitation. The final effluent, however, still contains some phosphorus depending upon the packing media and chemical used.

The discharge volume from Swan Brewery (prior to irrigation) is approximately 600 000 kL per annum. Of this 400 000 kL is used for irrigation of the factory area and 200 000 kL is discharged into the underdrainage area. The main concern regarding phosphorus is that it may leach into the groundwater from the underdrainage. The relevant regulatory authorities have indicated that the subsoil drainage phosphorus loads from the brewery should be in the range of 10-50 kg per annum (Evangelisti, 1994). This implies that the final effluent of 200 000 kL should contain not more than 50-250 µg/L phosphorus. Soil amendment or chemical precipitation alone have been shown, by the experimental results, to have the capacity to reduce phosphorus to this level. In either case, however, close supervision and/or a large amount of chemicals are required.

Phosphorus harvesting by planting grass (e.g. kikuyu/white clover grass) on the RMG amended soil and harvesting the biomass regularly would reduce the need to replace the

RMG, and should be a strategy considered seriously when determining which option to pursue. It should be noted that the adsorption capacity of RMG and red sand increases with time (Kayaalp, 1990; Wendy Akkers, 1995). Thus the above estimate for the need to replace RMG (based on short term column experiments) should be considered a conservative estimate. Amendment of RMG to a greater depth will also prolong media life. Doubling the amendment depth to 40 cm will double the life of the media (bed). There is, however, a practical limit to the depth of amendment. For harvesting with plants, for example, the depth of the root zone is generally limited to 30 cm, unless trees are used.

Chemical precipitation does not require land, however it requires a close control of pH and dosing of chemical. Regular analysis of phosphorus level in the wastewater and flocculation tests (jar tests) is essential to maintain a constant acceptable phosphorus level in the effluent. Sludge from the process, although less than 5% v/v, requires further treatment and disposal.

Since the sludge contains more than 95 percent of water content, with significantly low levels of phosphorus, the most economic method of treatment would be a sludge drying bed. This may be lined with coarse red sand to minimise contamination of ground water. Water in the sludge will be removed by infiltration and evaporation. Dry sludge will then be scraped off and may be used as fire break.

To ensure phosphorus discharge levels meet regulatory requirements, combined chemical precipitation followed by soil amendment may be appropriate, when practical/operational issues are considered. Soil amendment will reduce the risk of phosphorus level fluctuation by chemical precipitation alone. In turn, chemical precipitation prior to soil amendment will significantly decrease phosphorus and suspended solids level in the liquor. Infiltration rate and life span of soil media will thus be prolonged.

RECOMMENDATION FOR TREATMENT AND DISPOSAL

From the results of this experiment, the following recommendation can be proposed to the targeted industry, Swan Brewery.

Chemical treatment

The present wastewater effluent of 600 000 kL can be dividedinto two parts; 400 000 kL for irrigating the factory area and 200 000 kL to be treated before disposal. A diversion channel has to be constructed to separate the flow to direct 400 000 kL to the first pond and 200 000 kL to the second pond. To the stream of 200 000 kL chemicals will be added, firstly to bring the pH to the appropriate level, and secondly to precipitate phosphorus by chosen coagulant (such as alum). The chemicals get mixed well during the flow and are then allowed to settle in the second polishing pond.

The volume of approximately 5% sludge is pumped from the bottom of the pond to the sludge drying bed. The dried sludge can be disposed of as fire break material. The filtrate is to be directed back to the second polishing pond. The supernatant from the pond can be pumped to the disposal site for infiltration through the soil. Many types of grass such as kikuyu/white clover, Broomegrass and Ryegrass, which can remove 40-50 kg/ha/yr, can be planted at the disposal area. The grass uptake of phosphorus will protect for any inefficiency during the chemical process.

Soil amendment

From 200 000 kL of wastewater, 400 kg of phosphorus is removed by soil adsorption. The adsorbed phosphorus can then be removed by grass harvesting. Many types of grasses are available with the capacity to remove approximately 40-60 kg Phosphorus/ha/yr. It therefore requires 10 ha of land to remove 400 kg of phosphorus.

Red sand application to an area of 10 ha to a depth of 20 cm will provide capacity to remove phosphorus 400 kg/yr for five years. Any fall in growth of grass can be accommodated by this soil storage capacity.

CONCLUSION

1. Removal of phosphorus from wastewater, to satisfy the regulatory requirement, is expected to be achieved by chemical precipitation. The precipitate can be removed by sludge drying bed. Either aluminium sulphate or ferric chloride can be used as coagulant.

2. Chemical precipitation should be followed by some form of soil amendment, as the performance level of the chemical process can deteriorate. A minimum of 2 ha of land amended by red sand, or RMG with red sand, with appropriate grass is recommended.

3. Soil amendment alone can remove phosphorus but the soil is to be replaced periodically if the phosphorus is not removed by harvesting of grass specially grown in the area. An area of 10 ha of land with soil amendment of 20 cm depth by red sand can remove phosphorus for 5 years and RMG 10% with red sand can function for 8 years.

4. The need to replace the soil amendment can be significantly reduced (or eliminated) if appropriate grass, which can remove 40-60 kg Phosphorus/ha/yr, is planted in the disposal area.

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SLUDGE TREATMENT AND DISPOSAL CONSIDERATIONS FOR SMALL WASTEWATER TREATMENT PLANTS

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ABSTRACT

Although the proportion of solids in wastewater is very low, if not removed from small treatment plants there is a gradual accumulation over time. Many people tend to ignore the sludge treatment and disposal aspects in small wastewater treatment plants (WWTP). In some respects, this is due to ignorance for the assumption that there is enough capacity for the short term and that no problems will arise.

Small WWTPs (ie. <1000 equivalent population) generally operate on the extended aeration activated sludge principle which promotes considerable destruction of solids. Further destruction of organic matter generally occurs in either an aerobic digester, or in the anaerobic zone of a septic tank. In some instances, the partially stabilised sludge is stored in lagoons, where further degradation occurs and some separation of water results. In other instances, the sludge can be dewatered on drying beds.

One of the main issues with sludges from small wastewater treatment plants is compliance with Biosolids Management Guidelines. There is a requirement for an extended period of stabilisation which is not always easy to achieve. Furthermore, there is a requirement to analyse the sludge product and classify it for a particular range of uses.

The above requirements result in a limited number of disposal/reuse options for treated sludge from small WWTPs. This results in substantial unit costs. These need to be understood and considered when planning and designing small WWTPs.

A range of technical and cost issues for small, medium and large WWTPs are presented to enable a clearer understanding of sludge treatment and disposal criteria.

KEYWORDS

wastewater, sludge, biosolids, small WWTPs, pathogens, reuse, "secure" landfill, ATU, IDEA plant, facultative pond

INTRODUCTION

Raw wastewater, which is generally a combination of all household "liquid" type wastes and a small proportion of commercial and industrial wastewater, only contains an average of 300 mg/L of solids (ie. 99.97% of wastewater is water). Alternatively, this can be expressed as each person contributing approximately 65g of solids per day. One of the aims of wastewater treatment is to remove these solids from the wastewater and hence enable the save return of the treatment water to the environment.

The quantity of solids in raw wastewater is influenced by the:

- socio-economic level of the population;
- the proportion of commercial and industrial wastes;
- septicity of the wastewater;
- materials used in the wastewater collection network;
- level of industrial wastes control.

The solids in raw wastewater are principally of organic origin and they represent in the region of 85% of the total solids fraction. The remaining solids are inorganic and remain relatively unchanged during treatment. Similar influences to those listed above affect the proportion of organic matter in wastewater solids. Two additional factors are the type of soils prevalent in the catchment area and the degree of groundwater infiltration.

Through the wastewater treatment processes, the organic solids undergo various concentration and stabilisation stages. A large proportion of the organic solids are broken down by bacterial action to produce.

- more bacterial cells;
- other types of solids;
- water;
- gases (mainly CO_2 and CH_4;).

The aim of sludge treatment processes is to produce a stabilised material that can be utilised safely and to the benefit of the environment. If this is not possible, then the stabilised sludge should be disposed to a "secure" landfill type of facility.

When small wastewater treatment facilities (say <1000 equivalent population) are designed, the aspect of sludge treatment and disposal has traditionally been given little attention. It is generally considered that the natural degradation of solids that occurs in biological treatment systems and the relatively small quantities of sludge produced means that the operator will be able to cope in some way.

However, experience has shown that operators have not been supplied with adequate sludge treatment and disposal facilities to enable them to manage the treatment plant to a satisfactory standard. Furthermore, environmental standards for treatment plant operation have increased and National Biosolids ManagementGuidelines (ARMCANZ 1995) are being implemented. These mean that all facets of wastewater treatment need to be:

- of a high standard;
- reliable;
- consistent;
- auditable.

Generally, the resources available to undertake the abovementioned responsibilities are less readily available for small wastewater treatment plants (WWTPs) than for medium to large WWTPs. Designers of small WWTPs need to take this factor into consideration and endeavour to minimise costs and manning requirements. In some instances, a regional solution should be utilised. This paper elaborates on these with particular examples from the situation in Western Australia.

TYPES OF SMALL WASTEWATER TREATMENT PLANTS

There are numerous types of wastewater treatment plants used for small communities. From individual household WWTPs (eg. ATUs - Aerobic Treatment Units) serving up to 10 people, to activated sludge plants serving up to 1000 people, the per capita solids contribution is similar. Various sludge treatment and disposal arrangements are used, either aerobic or anaerobic or a combination of the two.

The various treatment plant types that will be discussed are as follows:

- Septic Tank Systems;
- Domestic ATUs;
- Package WWTPs;
- IDEA WWTPs (Intermittently decanted extended aeration);
- Facultative Pond WWTPs.

Septic Tank Systems

The standard Septic Tank System caters for up to 10 people and consists of two large settling tanks followed by an alternating leach drain system. The solids in the raw wastewater are allowed to settle over a 24 hour period in the two septic tanks that are connected in series. This results in up to 70% removal of suspended solids which accumulate principally in the bottom of the first tank.

Naturally occurring anaerobic bacteria decompose organic matter over time. A reduction in sludge levels and an increase in the floating scum layer result. The latter is a combination of grease and fats, plastics and floated matter from anaerobic digestion. An allowance is made for the volume of accumulated solids and scum in septic tanks.

A comprehensive study of Septic Tank Systems in Perth (Caldwell Connell Engineers) in the late 1970's to mid 1980's showed that Septic Tank Systems should be emptied of accumulated solids and scum at an average frequency of 4 years. This would ensure the longevity of satisfactory operation. More regular septic tank emptying is generally discouraged by the high cost of such an operation. In due course, this leads to total system failure and increased pollution load to the environment.

Accessibility to septic tanks for desludging is generally very poor and results in considerable disruption to gardens, wastage of time and unsafe conditions, with low hygiene standards used by workers.

In Perth the septage (septic tank system contents) is transported to a central purpose built facility in Forrestdale where after further stabilisation (lime addition), the solids are taken off site for disposal or composting and reuse.

Domestic ATUs

Serving up to 10 people, these systems utilise a series of septic tanks (primary) followed by a biological (secondary) stage of treatment and then disinfection of effluent. As in a standard Septic Tank System, solids and scum accumulate and decompose in the first half of the ATU.

Secondary treatment is by extended aeration (attached growth). Solids build-up from aerobic stabilisation of fine solids and dissolved matter is directed to the septic tanks.

The total build-up of solids in ATUs is greater than in Septic Tank Systems as the "secondary" sludge accounts for up to 40% of the total solids mass. Hence, ATU systems should be desludged at say 3 yearly intervals to ensure on-going satisfactory operation. An ATU system that is not fully loaded (ie. has less than 10 equivalent persons) will require desludging at longer intervals.

The same issues that apply to desludging septic tanks are relevant to ATUs.

Commercial ATUs.

WWTPs that treat flows from between 10 and 100 equivalent people and are an extension of the design approach used for domestic ATUs, are broadly classified as commercial ATUs. These units typically service commercial offices, light industry, group housing and holiday accommodation.

There are numerous variations in flow contributions for different applications. Similarly, there are large fluctuations in the mass of solids handled by these treatment systems. Most of the systems utilise the extended aeration attached growth principle and produce a high quality effluent. Sludge production is at a similar rate to that in domestic ATUs.

Most of the systems depend on occasional desludging by a "tanker" with disposal to a Local Authority septage disposal facility. Unfortunately these sites are generally poorly sited and managed and pose a significant pollution risk. However, most of these sites are being progressively decommissioned and alternative options, such as discharges to a Water Corporation WWTP, are being used.

A small number of systems use drying beds, with or without underdrains. Drying beds in some instances have proved to be very satisfactory, however on many occasions, they have proved to be a "headache" due to the substantial maintenance/ housekeeping requirements. In isolated locations, suitable sludge storage lagoons have proved to be acceptable. However, it needs to be remembered that regular cleaning of these lagoons is necessary and substantial costs can arise.

The use of a sludge storage tank offers considerable benefits in flexibility of operation and provides thickening. However, these are seldom installed because of the initial high cost.

Unless a commercial ATU is either "open-topped" or has easy access to the septic tank compartments, the monitoring of sludge levels is difficult and hence desludging is often infrequent (ie. process overload occurs due to inadequate solids settling).

"Package" WWTPs

These steel tanks were widely used in the late 1960's and the 1970's. No primary settling chambers were provided and the extended aeration process was utilised to produce a biological sludge, which was low in density and hence considerable in volume. Occasionally, an aerated thickening tank was provided prior to drying on sand beds but normally the biological sludge went directly to drying beds or a sludge storage lagoon. The whole approach was to delay any capital costs.

Stabilised sludge was either dried to a very friable state and set alight or mixed with the soil on the site. Because of the short life of these plants and their isolated location at the time this was an acceptable arrangement. The same would not be the case now!

EDEA WWTPs

Intermittently Decanted Extended Aeration (IDEA) plants are a modern version of the old "Package" WWTPs, although the IDEA units are not transportable as most of the treatment tanks are of concrete construction. No primary settling chambers are provided and all the sludge is derived from biological stabilisation.

Desludging is either manually or automatically controlled and occurs during either the aeration or after the settlement stages of the process. Because of the dilute nature of the sludge, liquid solid separation is undertaken in either an underdrained drying bed or a thickening/storage tank. Liquors are returned to the inlet of the plant.

Stabilised sludge can either be transported to another facility for dewatering and disposal or, after drying on-site, the sludge can be stored in an appropriate area and then classified to determine its end use. Once again, final disposal can only occur to a "secure" type landfill or the material can be reused.

Facultative Pond WWTPs

These low technology plants use the natural systems of sunlight, wind, bacteria and algae to progressively breakdown solids and purify the wastewater. Raw wastewater enters a large shallow basin (1.2m deep) through a submerged inlet pipe. The slow velocities allow very good settlement of solids and fines. Dissolved material is progressively stabilised by bacteria and algae.

With total detention times in excess of 40 days, the settled sludge undergoes natural anaerobic decomposition over many years. Once the sludge volume reaches a certain level, (generally > 10 years), the stabilised sludge needs to be removed. Ibis can be done in a variety of ways including:

- dredging the pond to an evaporation pan;
- dredging the pond to a mechanical dewatering machine (eg. centrifuge);
- dredging the pond to a tanker and then deliver to either a larger WWTP or to an injection machine;
- decanting the pond allowing sludge to dry and be removed by loaders and trucks.

This is a low frequency, but expensive exercise and needs to be considered at the initial design phase. The loading on the plant, site area and sludge reuse options will influence the methods employed.

GENERAL CONSIDERATIONS

Wastewater sludges, even after stabilisation, are quite objectionable and pose a significant health risk that needs to be managed. Besides the normal pathogenic bacteria, there is the risk of contracting disease if contact is made with sharp objects such as hypodermic needles. It is thus necessary to minimise operator contact with sludges and to utilise shredding equipment to reduce the size solids in the sludges.

With large numbers of small WWTPs the risks of infection of humans by wastewater sludges are high because it is difficult to satisfactorily manage safe and hygienic practices. In medium to large WWTPs sludge treatment and handling are largely enclosed and automated and operator contact is minimised. The number of personnel involved are very low and the system can be closely controlled with relevant training and surveillance.

Historically stabilised sludges from small WWTPs have been directed to the following range of routes;

- larger WWTPs
- septage TP
- burning onsite
- dispersal onsite
- removal by individuals and contractors for offsite reuse
- liquid sludge to septage tip
- dried sludge to landfill

Recently formulated National Biosolids Management Guidelines are being used by the Department of Environmental Protection to regulate the use and disposal practices of sludges in Western Australia. These guidelines require that stabilised sludges be;

- stored in correct manner;
- analysed for contaminants;
- classified for acceptable uses;
- tracked to final use;
- reused wherever possible.

If disposal to a landfill facility is the chosen option, then the landfill needs to be constricted and lined to a particular standard (ie. "secure"). If stabilised sludge is applied to land, then the receiving site needs to be assessed to ensure it can adequately "handle" the contaminant levels in the product.

The Guidelines endeavour to satisfactorily manage stabilised sludge to protect public health and the environment. This means that considerably more attention needs to be given in the DESIGN, CONSTRUCTION AND OPERATION AND MAINTENANCE of sludge treatment and storage facilities at WWTPs. The relative cost implications of the Guidelines on small WWTPs is several times greater than the impact on medium to large WWTPs.

To offset the cost of sludge treatment and disposal facilities for small WWTPs a regional approach should be considered. A regional sludge digestion, dewatering and storage facility could provide significant cost savings to small facilities. In addition, there would be reduced sludge handling requirements and overall reduction in safety risk.

CONCLUSIONS

The area of sludge treatment and disposal for small WWTPs has generally been inadequately considered in the design of these facilities. Designers have either decided that the issues are of negligible importance or that operators will be able to handle the situation.

Higher community expectations in the standard of WWT, the increased risk of disease transmission in handling WWT sludges and the evolvement of the National Biosolids Management Guidelines are forcing improvements in sludge management. This applies to small, medium and large WWTPs.

These improvements in storage, classification, tracking and reuse require considerably greater effort and higher costs. For small WWTPs, the impact is significantly more than for the larger WWTPs due to the economies of scale.

To minimise the cost of sludge management, the number of small WWTPs could be reduced, (ie. replaced by larger catchment systems), or sludge management could be undertaken on a regional basis. These decisions need to be considered now to avoid significant increases in the cost of individual small WWTPs.

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MANAGEMENT OF WASTEWATER SLUDGES TO MINIMISE RISKS FROM HUMAN PATHOGENS

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ABSTRACT

Sludges that are produced from wastewater that has human faecal input are a potential risk to human health because of the possible presence of human pathogens. However, sludges that do not have a high industrial input have very good beneficial re-use properties as soil conditioners and slow release fertilisers. Sludge management should take into account both the risks and potential benefits of sludges. Sludges produced by settling contain high numbers of a range of pathogens. Further biological treatments such as anaerobic digestion and composting produce a more stabilised product that is referred to as biosolids. These treatments reduce pathogen concentrations but some pathogens appear able to survive. We have found that two pathogens of particular concern are salmonellae and *Giardia*. Salmonellae appear to survive composting in low numbers and can survive for long periods. They can also potentially regrow in biosolids. We found that salmonellae regrew in stored biosolids after one year of storage and in soil amended with biosolids. *Giardia* are also of concern because they survived sludge treatments and biosolids composting in high numbers. However we found that *Giardia* cysts were reduced to below detection limits by 3 months storage after composting. From our studies recommendations for sludge management are that biosolids should be composted and that biosolids should be stored after composting before becoming available for re-use options in which humans are likely to be exposed. Storage may result in regrowth of salmonellae so monitoring for salmonellae should be carried out.

KEYWORDS

Biosolids, composting, *Giardia*, risks, *Salmonella*, sludge, storage

INTRODUCTION

Wastewater that contains human faeces is a potentially hazardous material as it may contain organisms that can cause disease. If sludge is produced by treatment of this wastewater then the sludge is also likely to contain disease-causing organisms. However it is not necessarily the case that the aim of sludge management should be to remove all risks associated with pathogens in sludge.

Removing all risks associated with pathogens in sludge would require that sludge be completely free of pathogens or contained in such a way that there was no human exposure to sludge. Very high temperature processes can achieve pathogen elimination (US EPA, 1992) but there are associated financial and environmental costs. Energy costs are high and air pollution needs to be contained. Effort can also be put into providing a site where sludge is completely contained. However there are again environmental and financial costs involved. In many situations sites where sludge could be completely contained are difficult to find. Although complete pathogen elimination or containment may be an appropriate option in some cases the costs involved in reducing the risks to such a low level may not be justifiable or desirable in other cases. It is also important that sludge is not unnecessarily treated as a hazardous product without consideration of the potentially beneficial properties. Sludge can be a valuable soil amendment that can improve soil structure and add nutrients to soil (Ross *et al.*, 1991).

It is therefore suggested that sludge management should involve balancing costs and benefits with risks. The health consequences associated with sludge should be placed in the context of other influences on health and risks to health. Sludge management should aim to reduce risks to an acceptable level rather than aim to completely eliminate risk.

A risk assessment approach can be taken to manage risks associated with pathogens in sludges. This risk assessment approach involves identifying the hazardous microorganisms in sludge, determining acceptable levels for these microorganisms and determining how these levels can be achieved. This approach is described below.

In this paper sludge which has been stabilised and considered for re-use is referred to as biosolids.

HAZARDOUS MICROORGANISMS IN SLUDGE

The organisms that are likely to be most hazardous in sludge are enteric pathogens as enteric infections generally result in the excretion of high numbers of organisms in faeces. Therefore the first step in hazard assessment is finding out which enteric pathogens are causing infections in the community. As an example the enteric pathogens which were reported to have caused disease in Western Australia in 1991 are shown in Table 1. The risks associated with these organisms are likely to be similar throughout Australia. Organisms that are not causing reported cases of disease are not likely to be of concern in sludge as their prevalence or health significance will be low.

The potential hazard is then mitigated by a number of other factors. Firstly the number of reported cases is important because if more people have the disease then more people will be excreting the organism. Table 1 shows the number of reported cases in Western Australia for the enteric pathogens. Another important factor is the number of organisms excreted by infected people. This is also shown in Table 1. Also of importance is the sensitivity of these organisms to wastewater and sludge treatment processes. This information is not available for most organisms but Table 1 shows an estimated time that pathogens can survive in the environment. Finally the level of potential hazard associated with these organisms is also related to the dose-response relationship. If low doses can result in infection then the organism is potentially more hazardous. The dose to cause an infection of 50% is also shown in Table 1. As described previously each of the microorganisms was then given a risk score by combining each of these different factors (Gibbs and Ho, 1993). The microorganisms are ranked in Table 1 with the highest risk scores at the top and lowest at the bottom of the table. A higher risk score means that there is a greater potential hazard associated with this microorganism if people are exposed to sludge or biosolids.

Table 1. Risk Scores, Number of Cases, Excreted Load, Persistence and Infectious Dose for Enteric Pathogens in Western Australia in 1991

Organism	Risk Score	Number of Cases*	Excreted Load#	Persistence#	Dose to cause 50% Infection+
Enteroviruses	7	232	10^7	3 months	3 to 47
Rotavirus	6	259	10^6	(3 months)	5
Adenovirus	6	206	(10^6)	(3 months)	<100
Hepatitis A	6	142	10^6	(3 months)	100
Salmonella spp	6	785	10^8	2 months	1x10[illegible] to 1x10[illegible]
Giardia	6	1014	10^5	25 days	35
Trichuris trichiura	6	127	10^3	9 months	10
Campylobacter	5	1579	10^7	7 days	3×10^9
Shigella spp	5	285	10^7	1 month	300 to 6×10^4
Cryptosporidium	5	273	(10^5)	(25 days)	(10)
Hookworm ova	5	125	10^2	3 months	(10)
Enterotoxigenic E.coli	4	27	10^8	3 months	(10^4)
Entamoeba	4	70	10^5	25 days	2 to 76
Hymenolepis nana (dwarf tapeworm)	4	178	100	(3 months)	(10)
Strongyloides stercoralis	4	132	10	3 weeks	(10)

*Anon, 1992
#Shuval *et al.*, 1986
+Calculated from models described by Rose and Gerba, 1991
Numbers in brackets were estimated from data for similar organisms

This information suggests that the organisms of greatest concern in Australian sludges are likely to be enteric viruses, *Salmonella* spp. and *Giardia*. *Trichuris trichiura* also had a high risk score but travellers and recent immigrants could account for most of these cases, where as the reported rates for other organisms are likely to indicate a larger number of unreported cases. It is therefore suggested that initially enteric viruses, *Salmonella* spp. and *Giardia* should be the focus of sludge management.

ACCEPTABLE RISKS

The acceptable risk approach is based on the idea that there is a threshold of risk below which people are indifferent to changes in the level of risk. One way of viewing this indifference is that people would not be willing to commit their own resources to reduce risk below this level. People find different levels of risk acceptable under different circumstances. As described by a Royal Society Study Group (1983) people generally find a risk of death of 1 in 100 unacceptable in essentially all circumstances. However sometimes people chose to be involved in activities with risks of death between 1 in 1000 and 1 in 1 000 000. It was found that few would take action at a risk of death of less than 1 in 1 000 000.

It is suggested that sludge management should aim to achieve acceptable levels of risk and that these need to be developed for pathogens in sludge.

There are a number of approaches that could be used for deciding acceptable levels of risk associated with sludge. As described previously (Gibbs *et al.*, 1997) one principle that can be used is that the risks of disease through exposure to sludge should be significantly less than risks from other sources. It is suggested that risks should be less than 10% of risks from other sources.

Once acceptable levels of risk associated with pathogens in sludge have been determined then acceptable concentrations of pathogens in biosolids can be calculated.

If there is to be unrestricted exposure to biosolids then it is suggested that acceptable levels of risk and acceptable concentrations of pathogens in biosolids should be determined based on children as the most at risk group. It is considered that children will be more at risk than adults with unrestricted exposure because they may ingest biosolids in their home garden and are more immunological susceptible to infectious diseases. Some young children exhibit 'pica' behaviour where they eat soil rather than accidentally ingesting it. Children exhibiting pica behaviour have been reported to commonly ingest 5 g or more of a soil per day (Calabrese and Stanek, 1995). Therefore an ingestion amount of 5 g has been used in the calculation of acceptable concentrations in biosolids to be made available for unrestricted use.

If the exposure is restricted so that there is no general public access to biosolids then it is suggested that an exposure level of 0.1 g could be used (US EPA, 1989). This amount could be ingested as a result of hand contact or incidental contamination of food crops.

Acceptable concentrations of *Salmonella* sp, *Giardia* and enteroviruses were calculated for biosolids suitable for unrestricted and restricted exposure using the procedure described by Hu *et al.*, submitted. Results are shown in Table 2.

Table 2. Acceptable Concentrations for Enteric Pathogens in Biosolids

Microorganism	Acceptable Concentration	
	Unrestricted Exposure (5 g ingested)	Restricted Exposure (0.1 g ingested)
Salmonella spp	0.04 in 1 g (1 in 30 g)	1.7 in 1 g
Giardia	0.05 in 1 g (1 in 20 g)	1 in 1 g
Enteroviruses	0.001 in 1 g (1 in 1000 g)	0.06 in 1 g (1 in 20 g)

The procedure used to calculate acceptable concentrations of pathogens in biosolids has been described previously for salmonellae (Gibbs *et al.*, 1997, Hu *et al.*, submitted). For biosolids to be made available for unrestricted exposure it was decided that the risk of becoming infected with salmonellae from biosolids should be less than 10% of the background rate for children in the 1 to 4 age group which is 4 in 10 000 (Anura and Hall, 1992). By using this acceptable risk level, an ingestion amount of 5 g, and the dose-response relationship described by Rose and Gerba (1991), an acceptable concentration in biosolids was calculated to be approximately 1 salmonellae in 30g of biosolids. In the case of restricted exposure a soil ingestion amount of 0.1 g was used and this resulted in an acceptable concentration of approximately 2 in 1 g.

In a similar way an acceptable risk was calculated for *Giardia* in biosolids to be made available for unrestricted exposure. In this case the level of infection rather than reported incidence of disease was used as the background rate. This is approximately 1 in 20 (Boreham *et al.*, 1981; Boreham *et al.*, 1981; Rendtorff, 1954). If 10% of this background rate is considered an acceptable risk associated with biosolids then the acceptable risk is 0.005. By using the soil ingestion rate of 5 g and the dose relationship described by Rose and Gerba (1991) then an acceptable concentration for *Giardia* in biosolids is 1 in 20 g. For restricted exposure an acceptable concentration was calculated to be 1 in 1 g of biosolids. Although these risk levels are suggested as values to be aimed for, present techniques used to monitor biosolids for *Giardia* do not distinguish between infective and non-infective cysts. In addition techniques do not have the sensitivity to detect 1 *Giardia* in 20 g of biosolids so this could only be achieved by analysing many smaller amounts of biosolids at very high cost.

Therefore assessment of whether this risk level is met in biosolids is not considered achievable at this stage.

Determining an acceptable risk level and concentration in biosolids for enteric viruses is difficult. Enteric viruses consist of a broad group of viruses including hepatitis viruses, rotavirus, adenoviruses and enteroviruses. For enteric viruses the reported annual rates of infection for the different types of viruses are low so it is suggested that the acceptable risk level of 1 infection per 10 000 people per year which was suggested by Regli *et al.* (1991) for drinking water could be used. It is also possible that enteroviruses could be used as model viruses for other enteric viruses. Using the acceptable risk of 1 in 10 000, the soil ingestion rate of 5 g and the dose response relationship described by Rose and Gerba (1991) for Echovirus 12, then the acceptable concentration of enteroviruses in unrestricted exposure biosolids is 1 in 1000 g. In the case of restricted exposure where the soil ingestion amount is 0.1 g then an acceptable concentration is 1 in 20 g. Again detection of enteroviruses in biosolids is technically very difficult and expensive. Therefore determining whether this risk level is met in biosolids is not routinely possible at this stage.

MANAGEMENT TO ACHIEVE ACCEPTABLE LEVELS OF RISK

As described above one way to manage risks associated with pathogens in sludge is to try to prevent exposure by containing sludge in a protected site such as a secure landfill. In this case levels of pathogens in the biosolids are not of concern.

For some other disposal or re-use options public exposure is also unlikely. For example when biosolids are used for land rehabilitation or forestry, where public access is restricted and where there is no run-off or leaching from the site, then pathogens should not be of concern.

However when there is a possibility of direct public exposure, and possible hand contamination or contamination of foods, then pathogen levels should not be higher than the acceptable concentrations described in Table 2.

Research has been carried out on ways of reducing pathogen concentrations in sludge and biosolids. Results of these studies and a comparison with acceptable risk levels are given below.

Sludge Treatment

Studies have shown that wastewater sludges produced by physical sedimentation processes contain high numbers of pathogens (Gibbs *et al.* 1995). Monitoring was carried out on sludges from three wastewater treatment plants in Perth and results are shown in Table 3. Mesophilic anaerobic digestion reduced pathogen concentrations but did not eliminate pathogens. Concentrations of all three organisms were above suggested risk levels for unrestricted exposure, and *Giardia* and enterovirus concentrations were above suggested risk levels for restricted exposure.

Table 3. Average Concentrations of Pathogens in Sludges from Three Perth Wastewater Treatment Plants

	Average Number of Organisms/g wet weight			
Type of Sludge	Primary	Activated	Digested	Dewatered
Microorganism				
Salmonella	45	1.6	0.46	0.62
Giardia	2200	1000	3100	820
Enteroviruses	110	86	19	3

Storage

Studies were carried out to see if storage would reduce pathogen concentrations to acceptable levels. Dewatered biosolids were stored for 60 weeks. Piles contained 40 m^3 of biosolids and were approximately 1 m high. Samples were analysed for the presence of *Salmonella*, *Giardia* and enteroviruses as described by Gibbs *et al.* (1995). Enteroviruses were not detected in 40 g samples. *Salmonella* and *Giardia* results are shown in Figure 1.

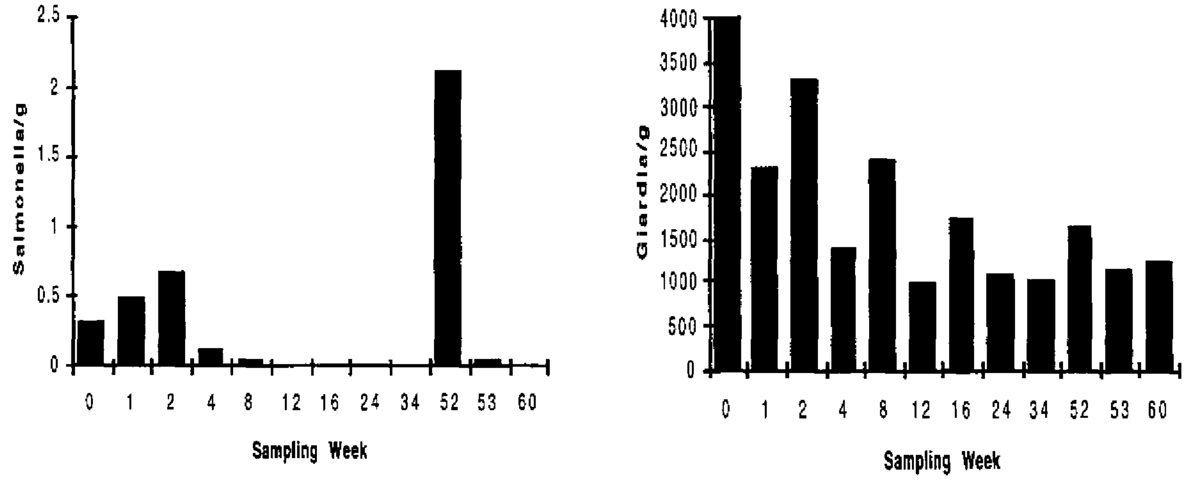

Figure 1. *Salmonella* and *Giardia* concentrations in stored biosolids.

Figure 1 shows that *Giardia* concentrations remained at greater than 1000/g after 60 weeks storage. It is not known whether these cysts were infective but in the absence of any suitable test for infectivity the possibility that some cysts are infective cannot be discounted. Also Salmonella appeared to regrow after 52 weeks storage. The possibility that there are infective *Giardia* present and that *Salmonella* regrowth may occur means that storage for only one year is not an adequate preparation for biosolids for restricted or unrestricted marketing.

Composting

Monitoring was carried out on compost produced from a mixture of biosolids, green waste and sawdust by two commercial composting companies. Composting was carried out for a 9 to 15 week period. A total of 18 windrows were analysed for *Giardia* as described by McInnes *et al.* (1996). All 18 samples contained *Giardia* at concentrations ranging from 5 to 6900 per g. The average concentrations for the two composting companies were 200 and 610 *Giardia* cysts/g. *Giardia* concentrations did not appear to be significantly reduced when the dilution is taken into account as dewatered biosolids contained an average concentration of 820 *Giardia* cysts/g and compost was produced by dilution of biosolids with 50 to 70% green waste and sawdust. A total of 11 windrows were analysed for *Salmonella* spp. as described by Hu *et al.* (1995). *Salmonella* spp. were detected in 7 out of the 11 windrows with an average concentration of 0.09 *Salmonella* spp. per g. *Salmonella* spp. concentrations therefore appeared to be reduced when compared to dewatered biosolids, which had an average concentration of 0.62 *Salmonella* spp. per g, although again some of the reduction would be due to dilution.

If *Giardia* cysts are considered to be infective then composted bisolids is not considered suitable for marketing. *Salmonella* spp. concentrations were higher than considered suitable for unrestricted marketing but less than the proposed risk level for restricted marketing.

Compost Storage

Monitoring for *Giardia* cysts was carried out on two piles of compost stored for 10 and 20 weeks after composting. Results are shown in Figure 2.

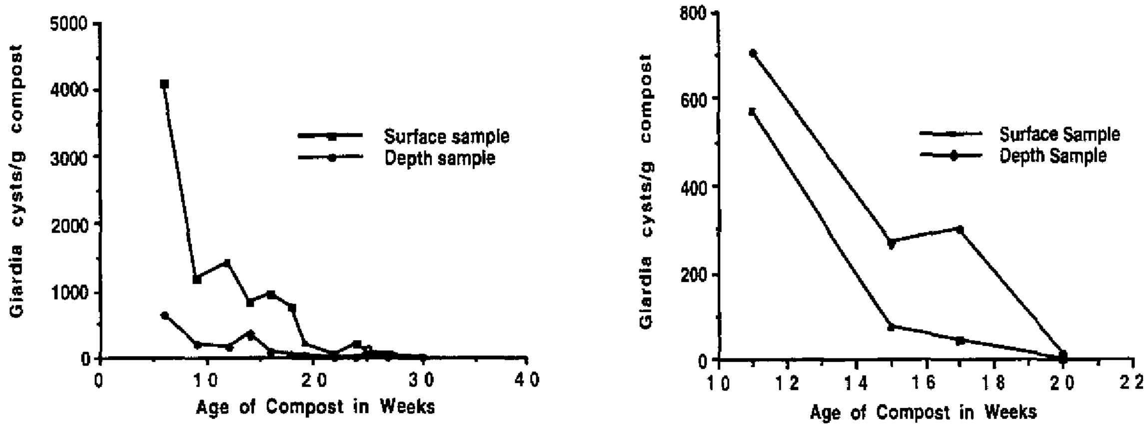

Figure 2. *Giardia* concentrations in two stored composted biosolids windrows.

Figure 2 shows that storage after composting resulted in a reduction in *Giardia* concentrations. After 20 weeks storage following composting *Giardia* concentrations were below the detection limit of the test. This is higher than the proposed limit so it is not known whether the risk limit was achieved after this amount of storage time. However it was shown that storage following composting appears to be an effective method of destroying *Giardia* cysts in compost and modelling could be carried out to determine what length of storage time could theoretically achieve the proposed risk level.

Soil Amendment

Dewatered biosolids was applied to soil and monitoring was carried out for 37 weeks as described by Gibbs *et al.* (1995). Enteroviruses were not detected and results for *Salmonella* spp. and *Giardia* are shown in Figure 3.

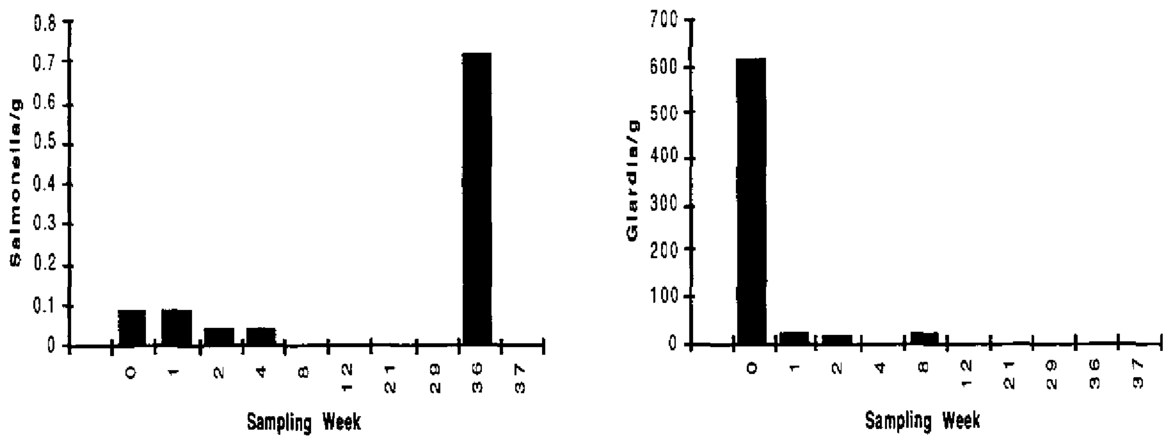

Figure 3. *Salmonella* and *Giardia* concentrations in soil amended with biosolids

Figure 3 shows that *Giardia* cysts were not detected in soil amended with biosolids after 8 weeks. However *Salmonella* appeared to regrow after 36 weeks storage. This shows that *Giardia* should not be of concern in soil amended with biosolids but regrowth of *Salmonella* spp. could be a problem.

DISCUSSION AND CONCLUSIONS

Risk levels have been proposed for *Salmonella* spp., *Giardia* and enteroviruses in biosolids but techniques are not sufficiently sensitive to detect *Giardia* and enteric viruses to these levels. Assessment of viability is very important for *Giardia* but viability assays are still in early development stages. Further work is needed on viability assessment and on improving the sensitivity of *Giardia* assays. The hazard assessment predicted that enteric viruses were the greatest potential hazard in sludge and biosolids but they were not detected in many samples. Biosolids appeared to have a toxic effect on culturing viruses in the laboratory so monitoring may have underestimated virus concentrations. Further development of techniques for testing viruses in biosolids would be of great benefit.

Studies of possible ways of achieving proposed risk levels have shown that storage and composting do not appear to achieve desired pathogen reductions. However an additional storage time following composting reduced *Giardia* concentrations to below the detection limit of the test. This is promising as a way of removing *Giardia* from compost. *Salmonella* spp. may still be of concern as *Salmonella* spp. regrowth occurred in stored biosolids and soil amended with biosolids. Also the regrowth potential of composted biosolids appears to increase with storage time (J. Sidhu, personal communication).

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REUSE AND TREATMENT SYSTEMS

FOREST IRRIGATION AND STORAGE POND PERFORMANCE IN COLD CLIMATE

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ABSTRACT

Wastewater from a small village in northern Sweden called Klovertrask was used for forest irrigation. The objectives were to reuse nutrients in the wastewater from the village, to avoid eutrophication of the recipient, and to improve the growth of the forest area. Wastewater was pumped from the village to a storage pond. During the growing season, of June to September, wastewater was pumped from the pond into the irrigation system.

The system functioned well during the first year of operation. However, there were some problems with the automatic regulation of the irrigation system. The removal of nutrients in the storage pond was effective, 60% for nitrogen and 65% for phosphorus. The nutrient load to the forest was 20 and 2 kg/ha for nitrogen and phosphorus respectively.

BACKGROUND

Wastewater irrigation systems are rare in Sweden. The system in Klovertrask, described in this paper, was the first one built in northern Sweden. Other existing systems are situated in the south of Sweden where the climate is warmer. On Gotland, an island situated near the southern east coast of Sweden, irrigation systems have been used, since the 1980s to utilise wastewater for irrigation of cultivated land areas. The primary goal is to reuse water. Most of the substances in the wastewater are removed in the pond systems before irrigation. The removal of phosphorus and nitrogen has been estimated to 70% and >90% respectively (Larsson, 1995).

In other systems wastewater is used to irrigate fast-growing trees (*Salix species*). Here the purpose of the irrigation is to supply nutrients as well as water to the trees. The trees are harvested and used for energy production. In one of the sites, Svalov, very promising results were achieved with respect to nutrient uptake by the trees (Hasselgren, 1999).

Reasons to use forest ecosystems as land treatment systems for wastewater were given by McKim *et. al.* (1982). Forest land is often less expensive than agricultural land, requires less management input, and is often more acceptable to the public because trees are not a part of the food chain. In addition, forest land represents a great potential for recycling municipal waste partly due to the highly permeable soils generally found in established forests.

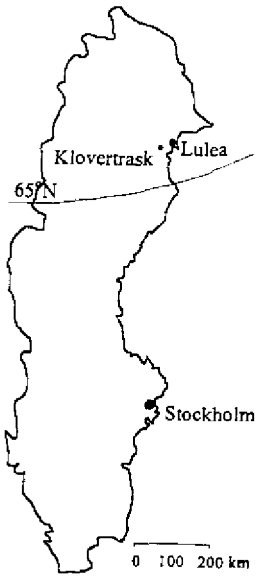

Figure 1.
Location of Klovertrask.

The storage pond included in the system at Klovertrask was not meant to work as a treatment unit. However, due to the long storage period the constituents in the wastewater will undergo physical, chemical and biological transformations. The main treatment processes of interest in ponds are: sedimentation of solids, anaerobic digestion of settled organic solids, aerobic stabilisation of organics, nutrient removal, and natural disinfection (Price *et. al.*, 1995). Due to the low water temperature and lack of oxygen and sunlight, the dominant process in ponds during the ice-covered period is sedimentation.

Major pathways for removal of nitrogen from wastewater ponds, as suggested by Reed (1985) are volatilisation of ammonia (NH_3) and benthic deposition of organic nitrogen. Other minor pathways were suggested to be nitrification-denitrification, algal uptake, and ammonia adsorption.

Phosphorus may be removed by benthic deposition, algal uptake, precipitation and adsorption of inorganic phosphorus (Ferarra and Harlemann, 1980)

THE EXPERIMENTAL SITE

Klovertrask is a small village in the northern part of Sweden with about 230 inhabitants situated 50 km south-west of the city of Lulea (**Figure 1**).

From the middle of the 70s until 1996 the wastewater from the village was treated in a mechanical-chemical treatment plant. During periods of high inflow to the plant, especially the snow melting period when the incoming flow can be up to ten times larger than the average flow, the treatment volumes of the plant were insufficient. During these periods the recipient for the treated wastewater, a small stream which was sensitive to nutrient loading, received a large amount of insufficiently treated wastewater.
A great deal of effort was put into a renovation of the sewer system, but did not result in a noticeably lower incoming flow.

During a few years there were discussions in the municipality of Lulea concerning a reuse of the nutrients in the wastewater. Forest irrigation using wastewater was one strategy that could be used in regions like northern Sweden, where the main part of the land area is covered with forest. The Swedish forest company, SCA, which owned forests around Klovertrask, agreed to dispose a forest area for wastewater irrigation.

Therefore, with the aims to reuse nutrients, to avoid eutrophication of the recipient, and to improve the growth of the forest area, half of the incoming wastewater flow to the treatment plant was used in the irrigation system. Only half of the flow was used was because of the high investment costs for the storage pond included in the system. The remaining part of the wastewater flow was still treated with chemical precipitation in the treatment plant.

A research project was planned to run during three years as a collaboration between Lulea University of Technology, the Swedish University of Agricultural science, Umea, the Swedish forest company SCA, and the municipality of Lulea.

OBJECTIVES

One objective of this work was to describe the system used for forest irrigation in Klovertrask. Another objective was to evaluate the performance of the storage pond included in the system. Transformation and removal of nitrogen and phosphorus species and removal of faecal indicator micro-organisms was considered to be of the most importance in the evaluation.

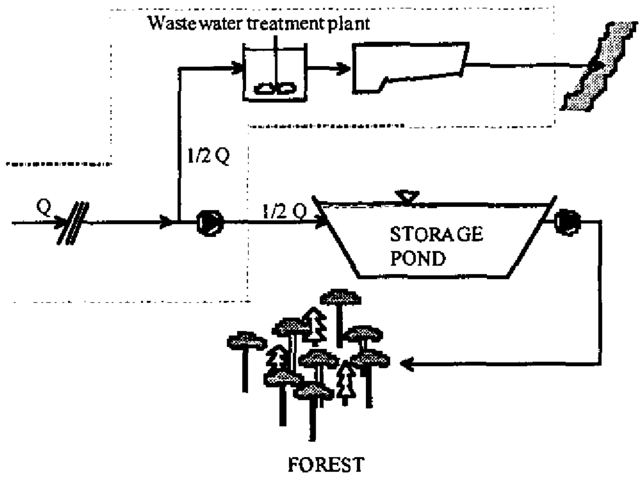

Figure 2. *The wastewater treatment system at Klovertrask.*

DESCRIPTION OF THE SYSTEM

The system was constructed in the summer of 1996.

Wastewater used for irrigation was, after screening in the old treatment plant, pumped 1.3 km to a storage pond near the forest area. This pond was used for storage since the water can only be spread during the growth season (approximately May-September). During this season water was pumped from a pumping station next to the pond into the irrigation system in the forest situated close to the pond. In the end of the irrigation period the pond was planned to be empty (**Figure 2**).

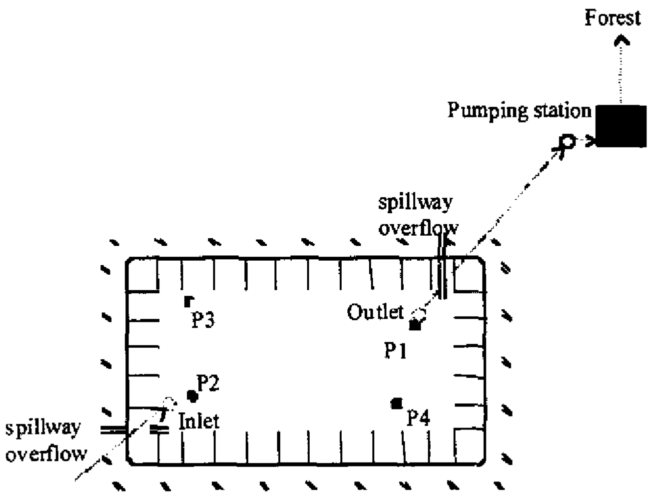

Figure 3. *The pond with sampling points.*

The pond

The pond (**Figure 3**) was approximately 45x95 m^2 with a depth of 4.3 m. The designed volume of the pond included a yearly precipitation of about 300 mm. The inclination of the pond walls was 1:2.

The ground of the area consisted of permeable soil and the groundwater surface had a high level. For these reasons the pond construction was made impermeable by a tight geomembrane and a drainage ditch was dug around the pond. From the pond two overflows were connected to the drainage ditch.

Around the pond a fence was raised to prevent animals from entering the water.

Wastewater entered the pond in a manhole 0.45 m above the bottom about 5 m from one of the corners. The outlet was located in the opposite corner of the pond 0.45 m above the bottom. The water flowed by gravity to the pumping station.

The irrigation system

The forest used in the project covered 8 ha and consisted of 15 years old Contorta Pines. Three irrigation methods were tested; one sprinkler system, one drip irrigation system, and one very simple system consisting of tubes with small holes. The whole distribution system was located above the ground level.
The irrigation system was fully automated.

To prevent clogging of the emitters in the distribution system the water was screened (0.1 mm) in the pumping station. The screen was automatically back-washed during operation.

Designed hydraulic loading during the irrigation season was 12 mm/week (ca 140 m^3/d). The water was spread during the night for around 10 hours. Under the assumption of 20% removal of nitrogen and phosphorus in the pond, the nutrient loading was estimated to be 40-50 kg/(ha.,year) and 10 kg/(ha.,year) for nitrogen and phosphorus respectively, calculated on the basis of average concentrations during the year in the inflow to the wastewater treatment plant. Two smaller areas in the forest (50x50 m^2) received a loading which were respectively two and three times as high as the standard loading. The purpose was to study if there were any seepage of nutrients from the ground to the groundwater/surface water in the case the loading was high and the trees were not able to assimilate all the nutrients. This study was conducted by the Swedish Agricultural University in Umea.

Costs

The total investment (construction) cost in the project was 4.8 million SEK.

Operation

In September 1996 the pumping to the pond started. All the incoming wastewater to the treatment plant was pumped to the pond. In the beginning of February 1997, the pond was full. Thus, the filling time of the pond was 144 days. During that time 17 200 m^3 of water had entered the pond. In one period towards the end of June 1997 another 1 100 m^3 of wastewater was added to the pond and in the end of September 1997 3 000 m^3 was added to the pond.

The average daily flow to the pond during pumping was 119 m^3. However, this value varied between 47 and 272 m^3/d. This corresponds to a daily per capita value of 500 L as an average.

The precipitation from September 1996 until October 1997 was about 700 mm, which meant that about 3 500 m^3 of precipitation entered the pond during that period. About 2 500 m^3 of water was estimated to evaporate from the pond during the same period, the evaporation was estimated to be 500 mm.

An ice cover started to develop on the pond in the end of October 1996 and the pond was covered by ice until the middle of May 1997. The maximum thickness of the ice was 63 cm, in the end of March. The ice was covered by snow, the thickness of the snow-cover varied between 10 to 35 cm during the winter.

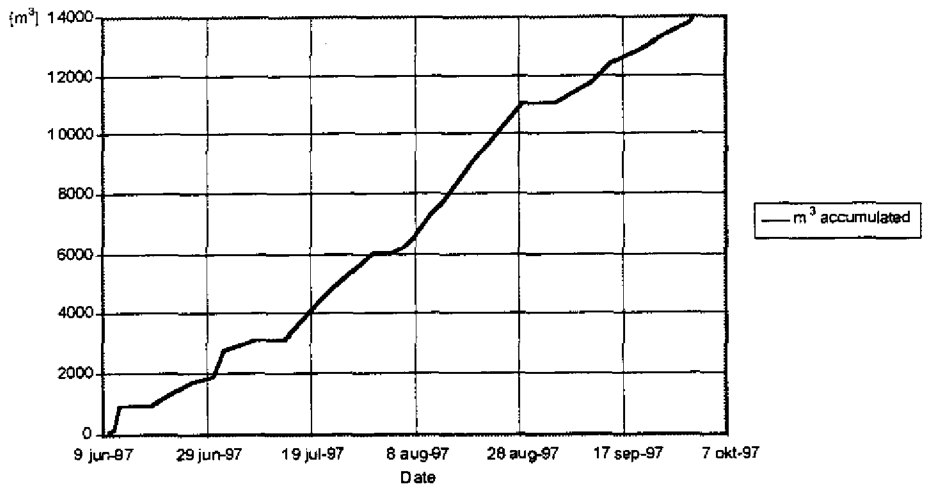

Figure 4. *The accumulated volume pumped from the pond into the irrigation system.*

In the beginning of June 1997, the irrigation started. The irrigation period ended in the beginning of October 1997. The accumulated volume pumped into the irrigation system is shown in **Figure 4**. Until the middle of July there were several problems with the pumping from the pond due to failure of the automatic regulation of the irrigation system. Twice during this period the automatic regulation was struck by lightning. In the beginning and in the end of August there were two failures of the automatic regulation.

The hydraulic loading in the forest during the periods, when the irrigation system functioned as anticipated, varied between 14-20 mm/week. The difference between the actual values and the designed 12 mm/week may be attributed to the intention of emptying the pond until the end of the irrigation season. This intention was not attained; when the irrigation period ended, the pond still contained about 7 000 m^3 of water.

CLIMATE DATA

The precipitation data for Klovertrask were obtained from SMHI (the Swedish Meteorological and Hydrological Institute), which has a climate station located in the village. Precipitation was the only parameter measured at the station. From July 1997 to September 1997, no data were obtained from the climate station in Klovertrask. During that period precipitation data were obtained from a climate station located 20 km from Klovertrask (Alvsbyn).

SAMPLING AND ANALYSIS

Investigations took place in the pond during the period 96/11/13-97/09/04. Transformation and removal of nitrogen and phosphorus species and removal of faecal indicator micro-organisms were of most importance during the investigations.

At every sampling occasion the water depth and the water temperature at every sampling point described below were measured. During the winter ice thickness was measured. When the pond was covered with ice, an ice drill was used to drill a hole for collection of samples. During the ice-free period a rowboat was used to reach the sampling points. No sampling was conducted from the middle of April 1997 until the beginning of June 1997 due to the ice break-up.

During the period of storage (September 1996 - June 1997) samples were collected once a month in the incoming water (as long as water was pumped to the pond) and at two points (P1 and P2, **Figure 3**) in the pond. At each point samples were taken at two different depths; around 50 cm from the bottom of the pond, and around 50 cm below the water surface (ice cover excluded). Temperature and pH were measured in the samples from each point. Total nitrogen, N_{tot} (filtered, f, and unfiltered, uf), ammonia nitrogen, NH_3-N (uf), nitrate nitrogen (including both nitrate and nitrite nitrogen), NO_3-N (uf) and total phosphorus, P_{tot} (f and uf) were analysed for every sample. For the filtration a 0,45 μm filter was used. For the analysis of faecal indicator micro-organisms one collection-sample containing equal volumes of water from the four sampling points in the pond and one sample containing incoming water was used. Micro-organisms analysed were heterotrophic bacteria, coliform bacteria, thermotolerant coliform bacteria and E-coli.

During the period of irrigation (June 1997 - October 1997) samples were collected at four points in the pond (P1-P4, **Figure 3**), at two different depths as during the period of storage. The lower sampling level at P1 represented the outflow from the pond, since the outflow was situated at this point. Around once every second week samples were collected for the measurements of pH and temperature and for analysis of faecal indicator micro-organisms. Samples for microbiological analyses were taken as two collection samples from the four sampling points; one sample from each sampling depth. Nitrogen and phosphorus species (as above) were analysed in samples, taken once a month, in P1-P4.

At a few sampling occasions the oxygen concentration in the pond was measured. A few samples were taken of the sludge at the bottom of the pond. The sludge was analysed for total phosphorus and Kjeldahl nitrogen.

Temperature was measured during sampling. The pH-values in the samples were measured around 2 hours after sampling. All samples for chemical analyses were stored in a refrigerator before analysis.

The preparation of samples for measurement of N_{tot}, NH_3-N, NO_3-N and P_{tot} were conducted according to Swedish Standards SS 02 81 31, SS 02 81 34, SS 02 81 33 and SS 02 81 02. The concentrations were then measured by using an automated procedure for analysis (an autoanalyser - TRAACS 800 - Bran+Lubbe).

The total phosphorus content in the sludge was measured by ICP. The Kjeldahl nitrogen analysis of the sludge was done according to Swedish Standard SS 02 81 01.

The accredited laboratory SVELAB conducted the microbiological analysis.

RESULTS AND DISCUSSIONS

In all diagrams shown, average values of the parameters measured were calculated for all the sampling points at each sampling depth. The differences between the values at the points at one sampling depth were very small at each sampling event.

Temperature

The results from the measurements during the sampling period are shown in **Figure 5**. The mixing of water in the spring turn-over is evident in the figure. The seasonal variations are also recognisable. In the beginning of September 1997, the whole water-body had almost equal temperature.

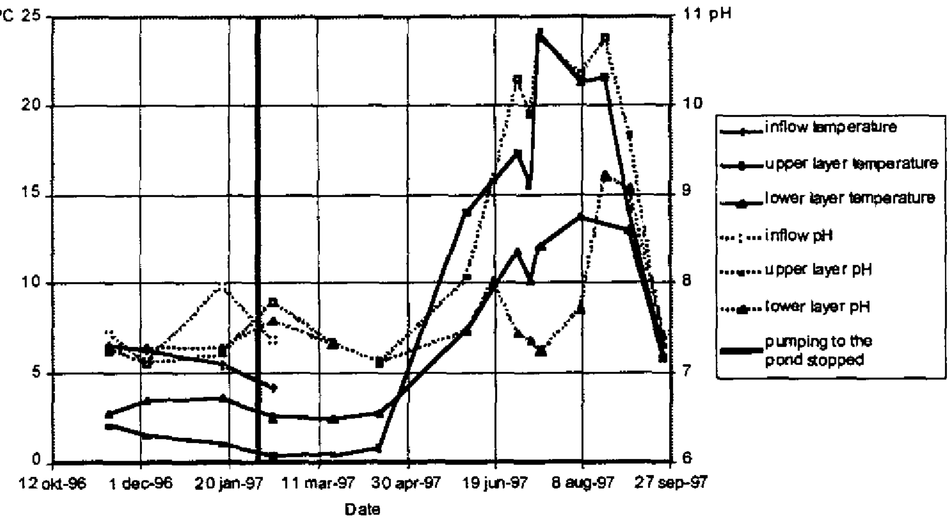

Figure 5. *Temperature and pH in the pond during the sampling period.*

pH

The pH-values are shown in **Figure 5**. The graphs shows that the pH in the pond, during the ice covered period reached a maximum value just after the pumping period was finished. After that the pH decreased, a fact that may be related to slow anaerobic degradation of organic matter. Eventually, the minimum values shown in the diagram were even lower. No samples were taken in the end of the winter when the ice was too weak to walk on. The impact of algal activity in the upper layer of the pond was obvious. It should be noted that the sampling occurred at daytime, the daily fluctuations of pH are not shown in the figure. In the middle of July, oxygen concentration measurements in the pond showed that it was only in the top layer (0-0.5 m) that the conditions were aerobic, a result of the algal activity.

Phosphorus

The concentrations of total phosphorus, unfiltered (uf) and filtered (f), during the sampling period, at the two sampling levels are shown in **Figure 6.**

A higher concentration in the lower layer during the winter, compared to the upper layer, was probably due to sedimentation, and the fact that the inlet for "fresh" wastewater, which had a higher concentration of phosphorus, was situated at this level. For nitrogen, the same phenomenon was observed, **Figure** 7.

In the end of the summer, a turn-over diluted the concentration of phosphorus in the lower water layer and the total concentration of phosphorus was equal in the whole water body.

The load of phosphorus into the irrigation system was estimated to be around 2 kg/ha (this could be compared to the assumed 10 kg/ha).

The total influent mass of phosphorus to the pond was estimated to be 60 kg, 16 kg of which were removed from the pond and transferred into the irrigation system. In the end of the summer the amount of phosphorus in the water volume of the pond was about 5 kg. The remaining part of the phosphorus, about 40 kg, thus was collected in the pond. The sludge in the pond contained 9 g P/kg TS, as an average of three samples. The thickness of the sludge layer was about 5 to 10 cm in the end of the sampling period, and the TS of the sludge was, as an average of three samples, 2%. The phosphorus content in the sludge thus was estimated to 40±20 kg.

Nitrogen

The total nitrogen (N_{tot}) and ammonia nitrogen (NH_3-N) concentrations in the pond are shown in **Figure** 7. The nitrogen was almost totally dissolved in the water until the beginning of the irrigation period. The dissolved part decreased during the summer, especially in the upper water layer. This could be explained by the algal activity. The part of NH_3-N of the total nitrogen content in the inflow, was about 70%. This part was higher in the pond, probably due to mineralisation and sedimentation of organic nitrogen. In the end of the storage period the

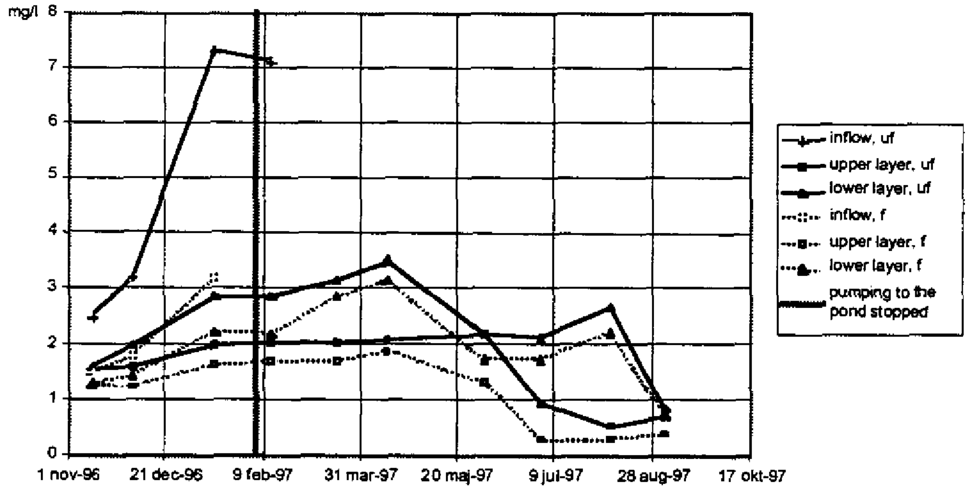

Figure 6. *Total phosphorus concentration, unfiltered (uf) and filtered (f) in the pond during the sampling period.*

NH_3-N-part was around 90%. This part decreased during the summer, and in the end of the sampling period almost no NH_3-N was present in the pond. The very low content of ammonia in the upper layer during the summer was explained by ammonia volatilisation as a result of the high pH in this part of the pond.

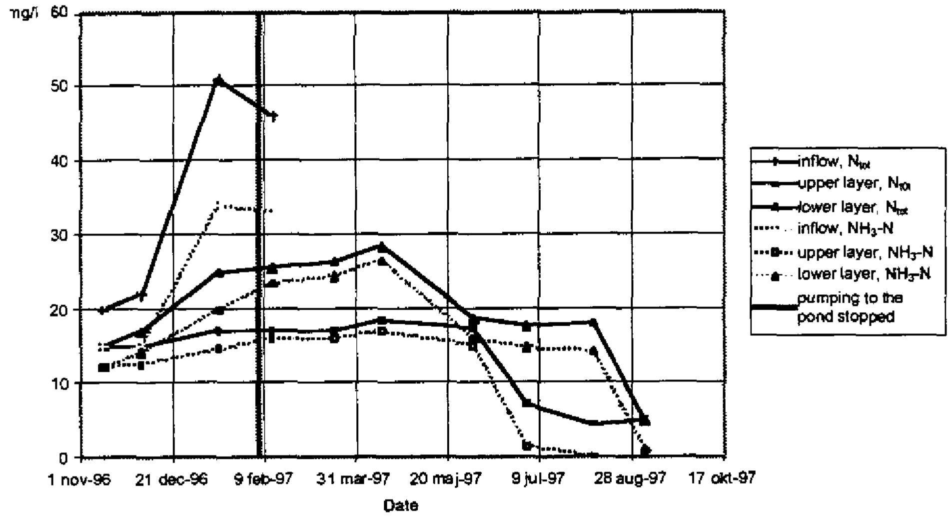

Figure 7. *Total nitrogen (N_{tot}) and ammonia nitrogen (NH_3-N) in the pond during the sampling period.*

The nitrate nitrogen did not exceed 0,2 mg/l during the sampling period. Thus, nitrification and denitrification was probably not a pathway for removal of nitrogen.

As for phosphorus, the figure shows, that the water body in the pond was totally mixed in the end of the summer.

The load of nitrogen into the irrigation system was estimated to be around 20 kg/ha (this could be compared to the assumed 40-50 kg/ha).

The total load of nitrogen into the pond was estimated to be 460 kg. 160 kg of the total load was removed from the pond into the irrigation system, and in the end of the summer the amount of nitrogen in the water volume of the pond was about 30 kg. The sludge in the pond contained 35 g N/kg TS, as an average of three samples. The thickness of the sludge layer was about 5 to 10 cm in the end of the sampling period, and the TS of the sludge was, as an average of three samples, 2%. The nitrogen content in the sludge was estimated to 170±70 kg. The remaining part of the total load, around 100 kg, was removed from the pond, probably by ammonia volatilisation.

Micro-organisms

The results from the microbiological analysis are shown in **Table 1**. The results show that the reduction of the analysed micro-organisms was relatively effective as soon as the pond became ice-free, in consequence of UV-disinfection from sunlight and biological activity (predation). The UV-disinfection was more effective in the upper layer as the light penetration was limited to the upper part of the pond. However, the number of the different bacteria varied a lot during the summer. This could to some extent be explained by the pumping of "fresh" wastewater from the treatment plant to the pond at some occasions during the summer.

Table 1: *Results from the micro-biological analysis*

Date	Sample point	Heterotrophs (2 d, 20°C) per ml	Coliforms (35°C) per 100 ml	Thermotolerant Coliforms per 100 ml	E-coli per 100 ml
96-12-04	Inflow	>100 000	>24 000	>24 000	>24 000
96-12-04	(c)	60 000	50 000	50 000	50 000
97-01-15	Inflow	-	>24 000	>24 000	>24 000
	(c)	-	>24 000	>24 000	13 000
97-02-13	Inflow	>100 000	>24 000	>24 000	>24 000
	(c)	36 000	>24 000	>24 000	>24 000
97-03-19	(c)	7500	>24 000	>24 000	>24 000
97-04-14	(c)	-	>24 000	>24 000	>24 000
97-06-03	0,5 m	300 000	260	112	90
	4 m	30 000	3 400	945	945
97-06-18	0,5 m	250 000	49	33	<2
	4 m	500 000	400	400	330
97-07-09	0,5 m	11 000	350	2	2
	3,5 m	110 000	350	2	2
97-07-16	0,5 m	42 000	23	23	23
	3,5 m	93 000	63 000	33 000	33 000
97-08-07	0,5 m	550	23	23	23
	2,5 m	30 000	>24 000	7 900	7 900
97-08-22	0,5 m	210	<1	<1	<1
	1,8 m	24 000	350	17	6
97-09-04	0,5 m	28 000	540	49	49
	1,5 m	22 000	16 000	16 000	16 000
97-09-23	0,5 m	90 000	1 300	240	130
	1,3 m	90 000	>24 000	490	490

c=collection sample from the four sampling points during the winter
- =no analysis

SUMMARY DISCUSSION

The separation of nutrients in the pond was effective. A higher nutrient load to the forest could maybe be achieved by changing the operation mode of the pumping to the pond. The pumping to the pond should start in the middle of the winter (January/February). During the snow melting period only a limited amount of water should be pumped to the pond. This operation mode will probably result in a continuos flow through the pond during the irrigation period, which will decrease the storage time in the pond, thus probably decreasing the separation of nutrients in the water before it reaches the irrigation system.

The main disadvantage with this mode of operation is that it could result in higher numbers of pathogenic micro-organisms irrigated into the forest. However, as the analysis of faecal indicator micro-organisms showed, the bacteria die-off seemed to be very fast as soon as the pond became ice-free. By increasing the detention time in the pond, for instance by using baffles, it would be possible to reach satisfying results, concerning the hygienic risks, in the water used for irrigation. Another disadvantage is that the problems with insufficient treatment volumes in the treatment plant during the snow melting period would remain.

CONCLUSIONS

The operation of the forest irrigation system has functioned well during the first year of operation. There were some problems with the automatic regulation of the irrigation system.

60% and 65% of the nitrogen and phosphorus respectively was removed in the pond. The nutrient load to the forest was 20 and 2 kg/ha for nitrogen and phosphorus respectively.

ACKNOWLEDGEMENTS

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GREYWATER REUSE FOR REVEGETATION IN ARID REGIONS

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ABSTRACT

Domestic greywater reuse is currently not permitted anywhere in Australia but is widely supported by the community, promoted by researchers, and improvised by up to 20% of householders. Its widespread implementation will make an enormous contribution to the sustainability of water resources. Integrated with other strategies in the outdoor living environment of settlements in arid lands the potential to improve quality of life is vast. This paper describes six options for greywater reuse under research by the Remote Area Developments Group (RADG) at Murdoch University and case studies are given where productive use is being made for revegetation and food production strategies at household and community scales. Pollution control techniques, public health precautions and maintenance requirements are described. The special case of remote Aboriginal communities is explained where prototype systems have been installed by RADG to generate windbreaks and orchards. Methodology and species selection used in design of the windbreaks and orchards, including 'sector', climate and soil analyses, to maximise shade, dust suppression and productivity are briefly outlined.

KEYWORDS

greywater reuse, revegetation, food production, remote communities, arid regions.

INTRODUCTION

The paradigm governing wastewater management has focussed on the pollutants in the wastewater and disposal as the solution. It relied on centralised water supply, sewerage and drainage systems with up to 85% of costs incurred in piping and pumping. This paradigm was developed on the Thames River in the last century and its appropriateness for the vast dry continent of Australia has been questioned (Newman & Mouritz, 1996) as has the transfer of these expensive centralised systems to developing countries (Niemczynowicz, 1993) and Australian indigenous communities (Race Discrimination Commissioner, 1994). Indeed, the arguments for abandonment of this paradigm in favour of one which cycles nutrients and resources for sustainability are perhaps now as evenly matched against the *status quo* as they were in the last century when the 'water carriage' lobby narrowly defeated the 'dry conservancy' lobby (Beder, 1993). The latter then also sought separation at source with reuse of dry and liquid products for agriculture although with much less scientific basis than what is available today. Goodland and Rockefeller (1996) proposed three general principles to enable the passage of the new sustainable paradigm: a) cease expansion of sewers and commence decommissioning them; b) promote on-site recycling systems that avoid pollution of water resources; and c) charge the true value of water. In Australia today there is little evidence that (a) is underway in urban centres; however (b) is well underway; and there is certainly discussion of (c) in the prevailing climate of economic rationalism. The focus of this paper is on-site recycling systems.

Reuse of wastewater occurs most effectively with on-site (localised) or small-scale treatment systems. A major study of Perth's wastewater management (WAWA, 1994) made it clear that it was not possible to reuse all the effluent from centralised treatment plants in the sewered suburban sprawl of Perth - there simply was not enough land for nearby broadacre application.

Thus to achieve the goal of total reuse the involvement of a local community in the urban situation would have to be be enabled and reuse options in the local context agreed upon. In sewered areas greywater reuse can still be implemented on-site. Greywater or sullage is effluent from the bathroom, washbasin and laundry, and for primary systems should exclude kitchen sink wastewater as it carries oils and high BOD. The more concentrated blackwater (from the toilet) can still go to the sewer along with kitchen effluent. In unsewered areas the blackwater can be treated separately or dry vault (pit or composting) systems utilised. Greywater reuse can result in cost savings (to both the consumer and state water authority), reduced sewage flows in sewered areas and potable water savings of more than 40% when combined with sensible garden design.

Significant impact on water and energy use might require greywater reuse to be coincidental with water-sensitive urban design, reduced lawn area, and possibly the growing of food at home and in public open space. There is immense community support for reuse of wastewaters (WAWA, 1994). This paper will review regulatory developments, describe six methods under research by the Remote Area Developments Group (RADG), present options for the three broad soil types in which trials are currently occurring and for remote Aboriginal communities, and explain the broader design approach that needs to be applied with greywater reuse.

CURRENT REGULATION

Domestic greywater reuse, governed by state and local government health acts, is currently not allowed in any of the Australian states although WA state authorities acknowledged that 20% of householders engaged in this practice in Perth (Lugg, 1994; Stone, 1996). In Queensland three options were developed for possible implementation (Department of Primary Industries, 1996). The model guidelines for domestic greywater reuse in Australia (Jeppeson, 1996) covered hand basin toilets, primary greywater systems (direct subsurface application) and secondary greywater systems (mesh, membrane or sand filtration prior to irrigation). For primary systems the guidelines have adopted the Californian approach requiring the use of a surge tank with a screen to remove lint and hair. Electrical power is therefore required for the automatic pump system and weekly inspection and clearing of the screen. The need for maintenance to these components by the householder resulted in some 80% of Californian systems being in an unsatisfactory condition. The recommendation of this approach as the solution for Australia is questionable. The updated standard AS1547-1994 guiding domestic effluent management (Standards Aust./NZ, 1996) is significantly more progressive in providing design criteria for a range of treatment systems with reuse and opening the way for further innovation.

Treated effluent from centralised plants is used on municipal ovals, parks and golf courses in many country towns of WA (Mathew & Ho, 1993). In New South Wales (NSW) treated effluent from centralised plants is allowed in urban areas (NSW Recycled Water Coordination Committee, 1993). National guidelines for the use of reclaimed water via dual reticulation have been prepared (National Health & Medical Research Council, 1996). The level of treatment recommended is secondary plus filtration and pathogen reduction. Alternatives to this include constructed wetlands which may achieve treatment equivalent to open water areas which will allow pathogenic die-off due to UV sterilisation.

In 1996 the WA Government released its Draft Guidelines for Domestic Greywater Reuse (HDWA, 1996) which allow the public to install greywater reuse systems in three shires as a means of conducting trials for 12 months (Fimmel, 1997). The three shires provided different soil types which would no doubt call for different design responses to pollution control and absorption: Bassendean (sands and coarse sandy clay); Kalamunda (shallow soil over rock in hills plus alluvial clay soils lower down on plain); and Kalgoorlie-Boulder (fine silty clay soils). Moreover many dwellings in these areas were unsewered. Funding would not be provided for the trials and the systems proposed would need to gain approval from the WA Health Department prior to installation. With the shire Environmental Health Officers as the public's first point of contact in seeking information and approvals they would need to receive comprehensive training.

The Western Australian State Government agencies quite rightly wants to move ahead and respond to the massive public interest in greywater reuse while at the same time exercising caution after the early Californian experience. The three shire trial will provide broad experience if a range of systems are allowed. The monitoring of these will provide invaluable information: Which systems are most appropriate for each of the conditons? How effective are the local government authorities in providing support and direction? How diligent are householders in maintaining these systems? What are the economic benefits? How effectively are greywater systems integrated into the landscape in relation to productivity and nearby recreation? What are the longer term effects on soil and plants? What is the nutrient balance between inputs, plant uptake, and percolation into the soil?

Experience does need to be gained for local conditons but there is a considerable body of literature for the trial shires to draw from. For example, there are McQuire (1995), Kourik (1995) and Ludwig (1994) for general interest while for contractors and do-it-yourself enthusiasts there are Jeppeson (1996) and Ludwig (1995). The design criteria provided in Standards Australia (1994) reflect the disposal paradigm while its revised version prepared with New Zealand authorities released as a draft only in 1996 allows for significant innovation.

SYSTEM CHARACTERISTICS

A greywater reuse system needs to protect public health, protect the environment, meet community aspirations and be cost-effective. Current on-site treatment systems have generally adopted the technology of the conventional activated sludge plant for large treatment systems. If removal of nutrients is required for installation of on-site units in nutrient-sensitive catchments, phosphorus (P) can be removed by alum dosing and nitrogen (N) by nitrification and denitrification in separate chambers or by intermittent aeration of a modified activated sludge set-up.

If the effluent is used for irrigation of garden plants there is the question as to why N and P should be removed. There may be an imbalance between plant requirement for the nutrients and the seasons, with a higher requirement in the warmer months than the colder months. Rather than removing the nutrients an alternative is to store the nutrients in the soil. Soils containing clay have the capacity to sorb ammonium and phosphate present in secondary effluent. Sandy soils can be amended with clay, loam or if convenient the 'red mud', bauxite-refining residue. The most progressive application of domestic greywater reuse appears to be in California. But even here the minimum prescribed depth of 430 mm for subsurface irrigation "ignore(s) the importance of aerobic bacteria and biota (found in profusion in the top few inches of garden soil) for digesting organic matter, nutrients and possible pathogens found in graywater" (Kourik, 1995).

SIX OPTIONS CURRENTLY UNDER RESEARCH FOR WESTERN AUSTRALIA

Amended Soil Filter

Fremantle Inner City Agriculture (FINCA) developed an 800 square metre community garden and is using the greywater from two adjacent houses to irrigate it. This is part of a water-sensitive, permaculture design approach which also involves harvesting rainwater from the two houses' roofs, heavy mulching and appropriate, low water use species selection for growing food in a perennial polyculture. Design and sizing of the system was generally in accordance with Standards Australia (1994) but performance monitoring and resident behaviour to date indicates the system is over-sized.

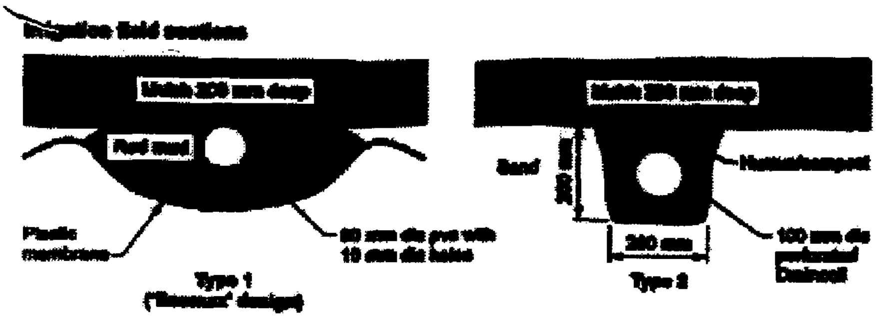


Figure 1: Cross-section of the amended soil filter

Greywater from the two houses enters a collection tank in the park by gravity. The duty field is a variation of the 'Ecomax' principle (Bowman, 1996) comprising two laterals of 20 m x 1.2 m and 25 m x 1.2 m wide. The plastic lined trenches are filled with a mix of 85% red sand and 15% red mud (with 5% gypsum in the latter to neutralise its alkalinity). The red mud and sand are by-products of bauxite refining to alumina. P is adsorbed into this clay material and N is removed from the system by intermittent drying and wetting causing nitirification-denitrification. Pathogens are filtered and die off. The field is heavily vegetated causing significant nutrient uptake and transpiration. Soil analyses to date indicate there is capacity for heavier liquid and nutrient loading and sludge build-up in the tank is negligable, i.e. application of AS1547-1994 design criteria resulted in over-sizing to the detriment of plant growth.

Sand Filtration

The Envirotech system consists of a receival tank where settling of solids occurs, a second chamber into which the effluent flows. When this is full effluent is pumped to the top of a deep-bed plastic-lined sand filter. Effluent filters to the bottom of this device under gravity and flows back to a third chamber of the tank, from where the treated effluent is pumped to the irrigation field. General practice is to chlorinate in this final chamber, although it may not be necessary for subsurface irrigation. Systems are being installed in NSW and Indonesia. A system based on Envirotech sand filtration for greywater reuse is now designed and awaiting installation by RADG at a WA site with Health Department approval.

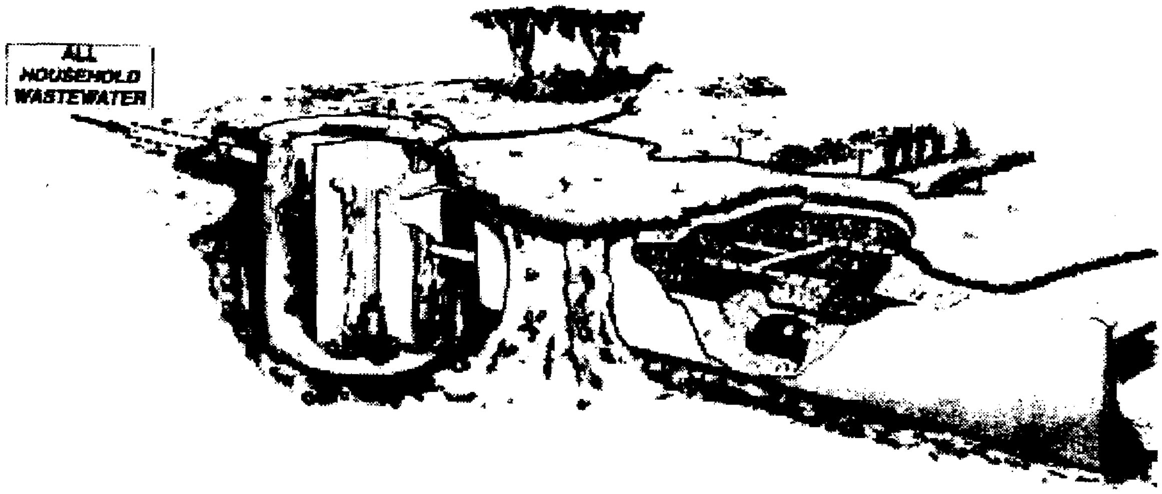


Figure 2: Diagram of the Envirotech sand filtration system

Wet Composting

The Dowmus vermicomposting toilet system can be upgraded to receive wastewaters - both blackwater and greywater (Cameron, 1994). In Canberra, ACT about 12 households have had trial systems installed for monitoring by Australian Capital Territory Electricity and Water (ACTEW) (Anon, 1996). Blackwater from the toilet enters a wet composting Dowmus tank and from there effluent goes to a second tank where greywater is also received. In this tank effluents are aerated around submerged volcanic rock media to achieve secondary standard treated effluent. From there the effluent goes to an irrigation storage tank in which chlorination occurs. The final effluent is mixed with rainwater to achieve further dilution and to improve the quality of water. Dowmus has been authorised to install five systems in WA for trial. One will be established at Murdoch University in the Environmental Technology Centre's permaculture system.

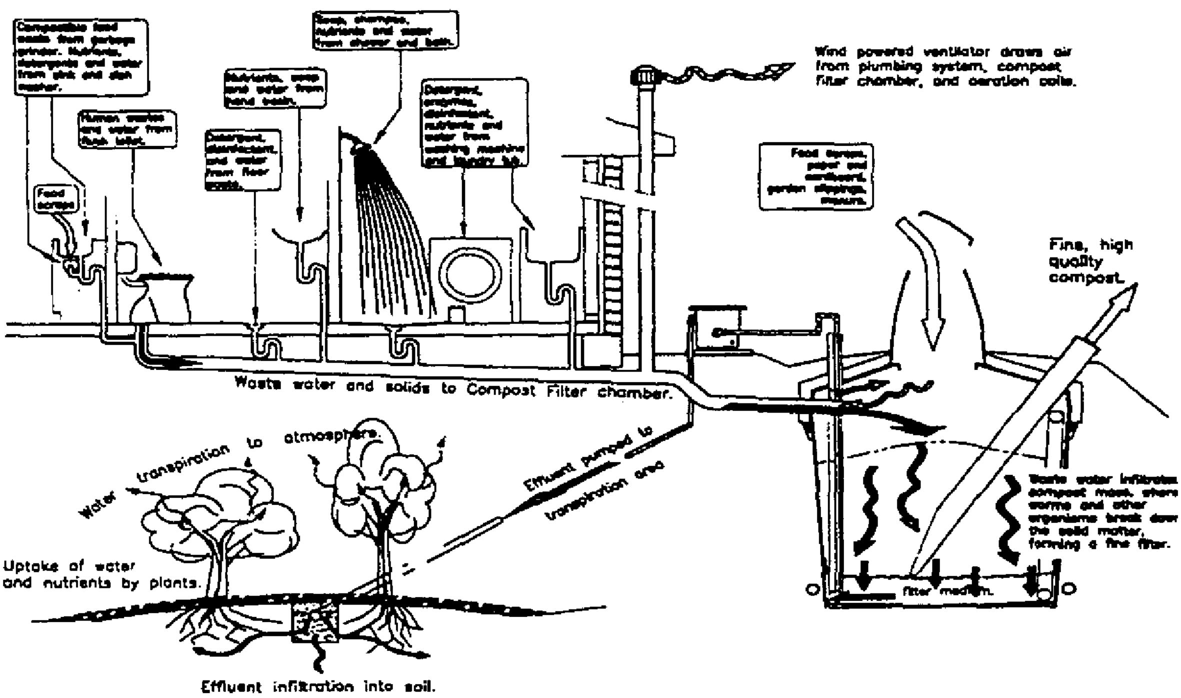
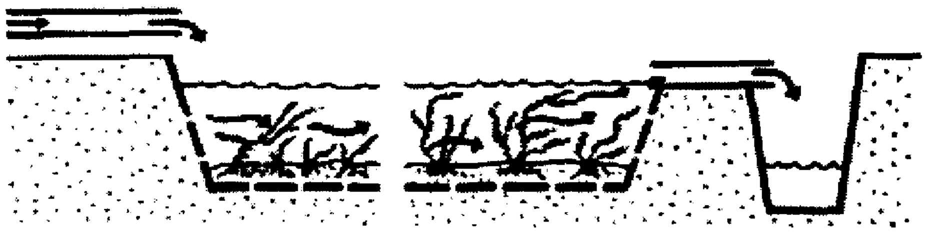


Figure 3: Cross-section of the Dowmus wet composting system

Constructed Wetlands

Mars (1996) is conducting a comparative trial on the effectiveness of the submergent aquatic plant *Triglochlin hueglii* and the emergent sedge *Schoeneplectus validus* in constructed wetlands for greywater treatment. Each of these species are reported to have a high ("luxury") nutrient uptake capacity. The former species is used in a surface flow wetland and the latter in a subsurface flow wetland. The aim is not only to verify treatment capability, but to use these local native species in a sustainable polyculture arrangement to produce food, thatching material, fodder and paper-making feedstock. Results to date are published in this volume.

a)Surface flow wetland (FWS)



b)Subsurface flow wetlands (SW)

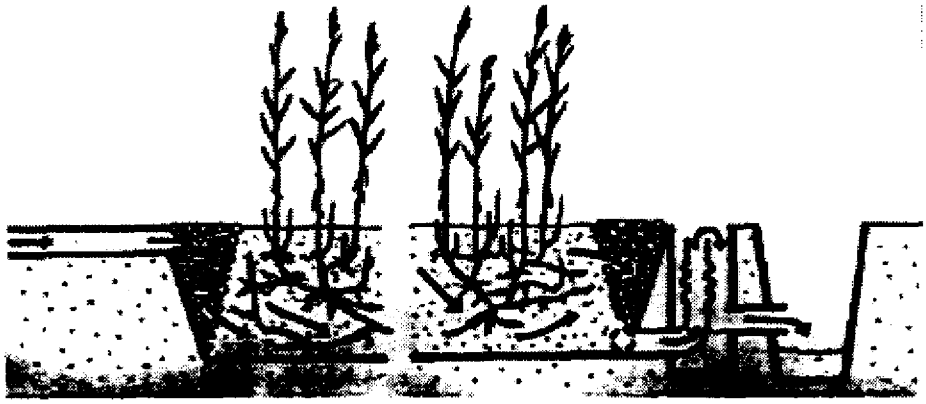


Figure 4 (a) & (b): Cross section of the surface and subsurface flow wetlands

Modified aerobic treatment unit

In Cottesloe, Western Australia, also a sewered suburb, a greywater reuse system that utilised the Biomax aerobic treatment unit was approved and installed in May 1996. Additional baffles have been installed in the anaerobic and aerobic chambers to enable more effective treatment of the lower biomass effluent input (c.f. combined blackwater and greywater). Effluent is irrigated to the front and back yards via 'Dripmaster' subsurface tubing. Monitoring is currently underway to evaluate the performance with the reduced biomass as a result of greywater influent only. Results to date appear to indicate that there is often not a significant reduction in BOD, SS and nutrient levels across the unit. There has been a similar experience with other aerobic treatment units. Research is being conducted to determine what improvements to the system design will be necessary.

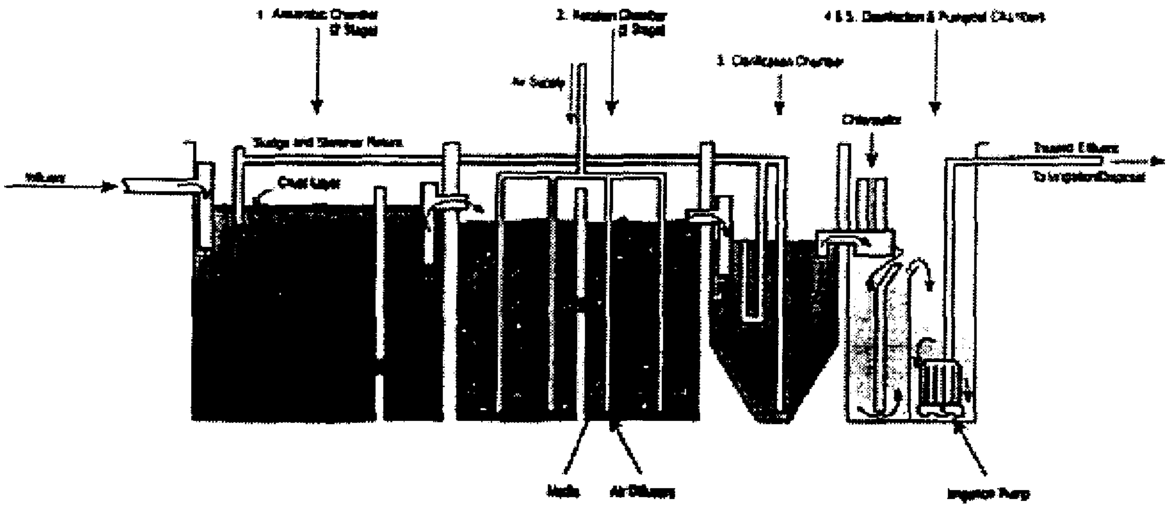


Figure 5: The Biomax aerobic treatment unit

Evapotranspiration systems
Evapotranspiration (ET) systems can be used in those areas where soil is comprised of more silts and clays and absorption fields have failed. These systems cost considerably less and require less maintenance than reticulated systems with lagoons (McGrath *et al*, 1991). Effluent disposal in the ET trench occurs primarily by soil evaporation and plant transpiration rather than soil percolation - as occurs in conventional leach drains. The trench essentially comprises a layer of gravel for distribution of effluent below a layer of river sand through which capillary action to the surface occured and in which plants grew.

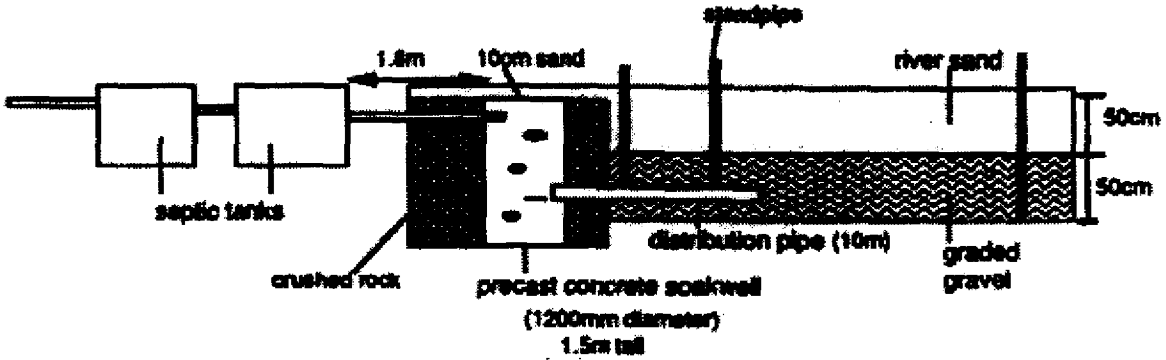


Figure 6: Cross-section of the evapotranspiration trench

ET systems receiving all domestic wastewater were installed in Aboriginal communities with community participation at Kawarra (McGrath, Ho & Mathew, 1991), Kalgoorlie, Irrungadji (McGrath, 1992), Halls Creek and Parnngurr School in the Western Desert. Systems in use in the clay soil shires of Perth including Kalamunda, were often inverted to some extent relying largely on evapotranspiration. The RADG ET system was developed to also improve the performance of these (McGrath, Ho & Mathew, 1990). Systems taking greywater only (alongside composting toilets) were installed at Tjuntjuntjarra in mid-1997 with a design intent of supporting native revegetation and orchards. In each case no problems were reported and in some, e.g. Halls Creek, vegetation planted on the fields is flourishing. Most importantly, no cases of ponding have been reported - one of the main reasons for developing this technology. It would be desirable, however, to conduct monitoring of performance over a longer period on these systems. They had generally been installed to the same size as leach drains to gain approval. Comparative monitoring with conventional leach drains would quantify the reduced size possible as a result of the better performance of ET systems and thereby reduce costs and improve irrigation of plants.

Ross Mars' system in the Perth hills suburb of Hovea referred to in Appendix 1 of HDWA (1996) is an 'absorption trench' that conforms with AS1547 and relies largely on evapotranspiration and to a lesser extent on absorption in the clay soil. The sand cover over the whole field of 7m x 7m interconnected piped trenches at 1500 centres is heavily vegetated with the high water demand plants sugar cane, banana, banna grass, canna lilly and vetiver grass. The system has performed satisfactorily without ponding since it was installed in 1994.

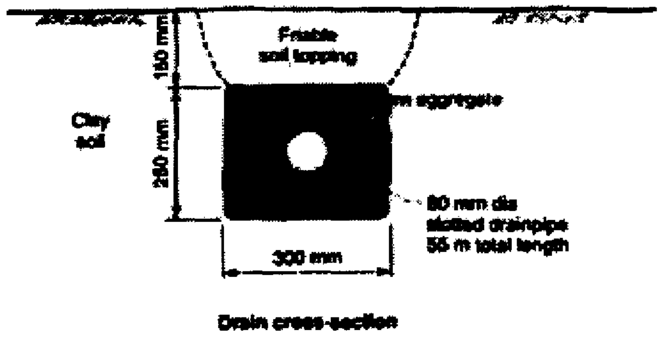


Figure 7: The Hovea absorption trench system

PROPOSED SYSTEMS FOR EACH SOIL TYPE

For cost-effectiveness and to most readily enable effluent reuse the followigng systems are recommended with respect to soil types:

Sand: *modified aerobic treatment units; amended soil filters; sand filters; constructed wetlands* (avoidance of groundwater pollution).

Clay: *evapotranspiration trenches* (avoidance of ponding)

Rocky slopes: *inverted evapotranspiration systems, sand filters* (avoidance of run-off)

The key problem to be overcome in each case is indicated in brackets. Wet or dry composting toilets can be used in conjunction with any of the above.

REMOTE ABORIGINAL COMMUNITIES

There are unique design considerations in the remote Aboriginal community setting. Unlike many of the large urban areas of Australia remote Aboriginal communities in arid lands do not have a diverse range of water sources. Typically there are groundwater sources whose sustainability in the face of growing populations is uncertain. At Coonana, for example, water shortages have been extreme (Race Discrimination Commissioner, 1994). However, there is poor public health in some communities and any reuse proposal needs to take serious consideration of this factor. Nevertheless, wastewater reuse can lead to improved public health. Separation of greywater and blackwater enables decreased loading on treatment systems and therefore results in greater reliability and performance. Dust control is accepted as necessary to alleviate disease, e.g. trachoma, which can be achieved through revegetation. Irrigation systems to establish trees use valuable potable water, are expensive and maintenance intensive. Greywater reuse evapotranspiration systems can be designed for low-cost, durability and low maintenance with sub-surface, gravity-feed, PVC piping.

Wastewater disposal systems often account for a major maintenance cost in remote Aboriginal housing and this is often because of poor initial construction by non-Aboriginal contractors (Pholeros, Rainow & Torzillo, 1993). A holistic response for on-site systems is necessary including separation of blackwater and greywater, use of evapotranspiration instead of absorption, interconnection of houses and systems to spread peak loads, back-up pit toilets to each house to cater for system failure, overcrowding and solids reduction, productive use of treated effluent, strict supervision of below-ground construction works, and effective management and maintenance.

In WA evapotranspiration systems are now fairly common in remote communities with tight soils since RADG commenced their implementation (McGrath, Ho & Mathew, 1991). Composting toilets have been installed at Wilson's Patch in the Goldfields and by Winun Ngari Resource Agency in the West Kimberley. Greywater reuse was recommended for Tjalku Wara in the Pilbara (Swanson, 1996) and a design using evapotranspiration was prepared by a regional permaculture practitioner. A trial greywater reuse system relying on evapotranspiration was approved for Frog Hollow in the East Kimberley (Kinnaird, 1997). However, the tendency has been to install deep sewerage to lagoons when funds become available rather than attempt to implement all of the above principles for a holistic response simultaneously. On-site and community-scale systems using one or more of the above six options need to be established in remote communities for research into their appropriateness and not just their technical suitability. In most cases, however, evapotranspiration systems will be appropriate and these can be adapted for simpler greywater reuse in parallel with blackwater septic systems or dry vault toilets.

Studies were completed for wastewater reuse from lagoons at Warralong and Jigalong in the Pilbara (Mathew & Ho, 1993). There was insufficient wastewater produced for irrigation of a football oval. Groundwater recharge was an option. The most suitable options were revegetation, orchards and vegetable gardens by subsurface or drip irrigation. If surface irrigation was proposed some form of disinfection to eliminate pathogens and enclosed storage to eliminate algae would be necessary. Reuse direct from lagoons could be subsurface from the overflow after the last lagoon or pumped from the lagoon to storage for later irrigation.

HOLISTIC DESIGN

Many concerns have been raised in relation to widespread implementation of greywater reuse without proper management or maintenance: reduced sewer flows, higher concentrations at treatment plants, public health risks, groundwater contamination, mosquito breeding, flooding during winter rainfall, sludge build-up and blockages. However, there is another issue for concern that may lead to some of these problems and others indirectly: poor design (or no design). Not just the design of the system itself but the manner by which the system is integrated into the landscape. Australian standards such as AS 1547 do, for example, specify minimum setbacks from houses and lot boundaries, provide ways of avoiding inundation and give design criteria for terraced disposal fields on slopes.

There are very few practical design methodologies that may serve the case of placement of a greywater recycling system in the house yard or community landscape. Two examples are:
* *hydroscaping*(Colwill, 1996) for sustainable garden aesthetic design; and
* *permaculture* (Mollison, 1988) for sustainable food production system design.

Hydrozoning will allow the placement of the greywater system in accordance with a garden layout designed for aesthetics and plant groupings of similar water needs.

Permaculture draws on a wider range of design tools including *zoning* for energy efficiency and *sector analysis* of the natural elements affecting the site (sun, wind, fire, view). Zones 1 to 5 in permaculture refer to areas of planting types (intensive salad beds, low maintenance orchards, through to natural bushland) placed in relation to house or settlement according to frequency of visits. Design with sectors allows the appropriate placement of windbreaks, shadetrees, water tanks, zones and other elements in the landscape.

The use of a design approach prior to installation enables placement of the greywater system in a landscape with respect to the vegetation type that it will support and its position in relation to other elements and natural influences on the site. If such considerations are ignored with a focus merely on the technical design of the system itself then improper management and maintenance and poor performance may still be the longer term outcome.

CONCLUSIONS

For the urban village, small country towns, or group housing a greywater reuse system utilising secondary treatment and disinfection maintained by a supplier may be most appropriate. For on-site greywater recycling at individual houses in a low-density settlement or remote community a primary system with large diameter subsurface irrigation 300 mm below the surface is appropriate using evapotranspiration in soils of low permeability. Filters, pumps and treatment units should be avoided as these may not be adequately maintained by the owner/occupier. Reuse from lagoons is commonly practiced in WA country towns. If nutrient removal is necessary a treatment system such as Aquarius or Ecomax with sufficient vegetation to utilise the nutrient is ideal. Data-gathering on the long term effects of greywater on plants and soils and their nutrient uptake capacity is necessary. Field trials are necessary to optimise the irrigation fields for plant growth, particularly in the case of food species. Evapotranspiration systems, for example, have typically been designed too deep in the past for this purpose. A standard code of practice on greywater reuse should be adopted. If managed correctly wastewater reuse in remote Aboriginal communities can not only result in water savings but also improved public health through dust suppression from revegetation, improved nutrition from locally grown food, and less system failures from decreased loading on treatment systems.

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GREYWATER VERSUS EFFLUENT REUSE - WHICH OFFERS MORE BENEFITS?

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ABSTRACT

Over the last decade more emphasis has been placed on water conservation, principally due to the increasing scarcity of new water supplies and the relatively high cost of those supplies. One avenue of reducing water use in the community is for householders to recycle all or a portion of their greywater (i.e. all household water used with exception of toilet water which is generally referred to as blackwater).

In Perth, during summer over 40% of total household water use is attributed to watering of lawns and gardens. If some of this demand could be replaced by the use of greywater an overall saving in water use would result. The two principal sources of greywater in homes are derived from the bathroom(s) and laundry. Approximately 40% of all in-house water use occurs in the bathroom(s) and laundry and this water tends to be the least contaminated of all greywater.

There are several concerns with greywater recycling that are not generally well appreciated. One is the fact that greywater does contain nutrients which over time contribute to the pollution of nearby surface and ground waters. Not all the nutrients are taken-up by the vegetation in the disposal field. Secondly, water demand in the garden/soil is seasonal and an alternative disposal/use is required during wet weather. This negates the apparent cost benefit of smaller sewers with greywater recycling.

An alternative water resource conservation approach is to reuse treated wastewater from neighbourhood and larger wastewater treatment plants. The Water Corporation provides treated wastewater to over 35 effluent reuse schemes in the Country area of Western Australia. This practise has been in place for several decades and serves the needs of country communities very well.

Feasibility studies have been undertaken for provision of treated wastewater for parkland and golf course irrigation at Mosman Park and a study is underway for provision of process/cooling water for industries in Kwinana. Both of these projects would utilise many million litres of effluent daily and this would be done under well controlled conditions that minimise health and pollution risks.

This paper examines benefits of greywater recycling and effluent reuse and details the advantages of each. A clearer understanding of this issue will result.

KEY WORDS

greywater, reuse, recycling, treated wastewater, effluent, neighbourhood, wastewater treatment plants, nutrients, groundwater

INTRODUCTION

Australia is a dry continent and the majority of the interior has a scarce supply of water. This is exacerbated by high evaporation rates due to the hot conditions.

Historically, Australians have adopted ornamental gardens that require significant quantities of water, particularly over the long, dry summer period. This is a legacy of the English settlement days many years ago.

Over the last two decades, Perth, the capital city of Western Australia, has experienced below average rainfall. In addition, the community has started to challenge the building of new water sources in the hills area to the east of Perth. This has resulted in a drive to reduce per capita water consumption, either by use of more efficient appliances and practices or the substitution of some of the water by other sources of water e.g. greywater, groundwater and treated wastewater.

One avenue of reducing water use in the community is for householders to recycle all or a portion of their greywater (i.e. all the household water used with the exception of toilet water which is generally referred to as blackwater). The two principal sources of greywater in a household are derived from the bathroom(s) and laundry. This accounts for approximately 40% of all in-house water use and this water is the least "soiled" of all greywater.

In Perth, during summer over 40% of total household water use occurs outside the home, on gardens. If some of this demand could be met by the use of greywater an overall saving in potable water requirement would result. There are numerous variations in the type of greywater recycling schemes but most are centred about recycling through toilet cisterns and the garden.

There are several concerns with greywater recycling that are not generally well appreciated. One is the fact that greywater contains appreciable levels of nutrients which over time contribute to the pollution of nearby surface and groundwaters. Where greywater is applied to the soil only a portion is absorbed by vegetation. Secondly, water demand in the garden is seasonal and an alternative disposal system is required during wet weather (i.e. >3 months of the year). This negates some of the apparent cost benefit of smaller sewers with household greywater recycling.

Greywater recycling is regulated by the Health Department to ensure there is a high standard of protection to the community in terms of health risk. When greywater recycling is proposed for deep sewered houses the Water Corporation needs to be consulted. Generally, greywater recycling in sewered areas is not approved because of the detrimental impacts on the wastewater conveyance and treatment systems. These impacts include lower flushing velocities in sewers and higher strength wastes that result in production of more toxic and odorous gases and more difficult and higher cost of treatment.

Two main options exist for greywater recycling in households, requiring either treatment or no treatment. In both cases application below ground is preferred, with this being a mandatory health requirement for untreated greywater in Western Australia. The biological treatment of greywater can be difficult to sustain over a prolonged period because of the low organic and nutrient content of the waste. The application of untreated greywater below ground also tends to be a problem over time. All these issues add to the cost of recycling greywater.

An alternative to onsite greywater recycling for conservation of water resources, is to extend the use of treated wastewater reuse. Whether wastewater treatment plants (WWTPs) are small or large there is some potential for reuse of the final effluent. Of the 80 country Western Australia Water Corporation WWTPs almost half the schemes recycle all or a large portion of the treated effluent. This practise has been in place for several decades and has provided these communities a valuable water source for their parks and gardens.

In the Perth metropolitan area an increasing quantity of treated wastewater is utilised on the wastewater treatment plants for cleaning, irrigation and spray purposes. In the Water Corporation's Wastewater 2040 Community Consultation Program (completed in 1995) there was a strong community preference for more widespread effluent reuse. This can be for groundwater recharge, saltwater intrusion control and industrial process/cooling water.

Feasibility studies have been conducted for provision of treated wastewater for parkland and golf course irrigation in Mosman Park, the Subiaco area and for provision of process/cooling water for industries in Kwinana. These schemes would utilise in excess of 15 million litres of effluent daily i.e. equivalent to 70 000 households or 0.2 million people. These projects would be operated under well controlled conditions that minimise risks to health and pollution.

Further investigations and communications of results are necessary in order to obtain sufficient information to determine the overall community benefits of various wastewater recycling/reuse schemes. Details of the various options need to be presented.

DISCUSSION

Greywater recycling schemes are prevalent in a small percentage of Perth households. In country Western Australia the schemes are more prevalent with some areas having up to 1 in 3 households with some form of greywater recycling. This is no doubt due to;

- hotter climate
- scarcity of water
- higher cost of water
- use of onsite WWTPs (i.e. "protection" of effluent disposal field)

Within deep sewered areas only trial greywater recycling systems are approved. However, it is understood that there are numerous unapproved schemes operating. In Perth metropolitan areas where deep sewerage is not available or proposed in the near future greywater recycling schemes are approved. This is the monitored by the Health Department of Western Australia, who also liaise closely with officers of the Water Corporation.

The various types of greywater recycling schemes include the following:

- manual (no treatment) - use of buckets/hose
- semi-automatic (no treatment) - shallow buried hose
- settling tank and buried infiltration system
- settling tank and in-roof storage for toilet use
- settling tank and filters - in-house or out-house use
- treatment plant (ATU) - in-house or out-house use

Above ground use of untreated greywater is not permitted because of health concerns, especially for young children, and possible overland flow across a property boundary. This method can also produce odours from decaying organic matter and attract flies and other insects. One of the big concerns is that it is difficult to control the presence of pathogenic bacteria originating from faeces soiled clothing. Constant application of greywater in one area can also affect infiltration rates due to matted hair and soap and detergent remnants.

Below ground application of untreated greywater is permitted because it removes the direct link between humans and the potentially possible pathogenic bacteria laden greywater. This method also generally ensures that plumbing connections are permanent whereas this is less likely in the above ground approach. As there is a need to protect public health all plumbing must comply with the Plumbing Code. Untreated greywater contains hair, lint, soap and detergents and over time these build up and clog infiltration holes and soil pores. A filter can be used to reduce this problem but regular and satisfactory cleaning in a safe manner is necessary. This does not always occur!

The use of settling tanks prior to a below ground infiltration system is beneficial but has little impact on clogging and the passage of nutrients to the groundwater. Filters do prolong the longevity of satisfactory operation of a system.

Full treatment of greywater does ensure a high quality effluent that can be used for a wide range of applications, from above surface spraying to toilet flushing. However, these individual type schemes are reliant on diligent servicing to ensure a high standard of performance. For this reason the preferred approach is for subsurface application such as drippers or infiltration drains. Other concerns with full treatment of greywater include the following:

- high cost of systems (>$5000)
- more complex systems
- longevity of reliable operation on greywater only
- sensitivity of system to soap and detergents usage
- removal of nutrients requires even more complex treatment plant

Use of treated greywater in toilet cisterns does provide a regular demand but WWTPs malfunctions can result in coloured and solids laden water from time to time. This is not acceptable to most householders. The need for a ceiling storage tank and the associated plumbing adds a significant cost to the system. Furthermore the controls on the tank can malfunction and cause problems.

Effluent reuse can be utilised from small to large WWTPs and is practised to varying degrees within the Water Corporation, a statewide organisation. The various alternative effluent reuse schemes in which the Water Corporation participates include the following :

- cleaning and process water onsite within large WWTPs (up to 3% of flow)
- irrigation of parks and gardens in Country towns (up to 100% of flow for over 35 towns)
- irrigation of tree plantation (100% of Albany flow)

- irrigation of vineyard (100% of Mt Barker flow)
- irrigation of parks, and race and golf courses (20% of Geraldton flow)

Other effluent reuse schemes being investigated include:

- irrigation of parks and golf course at Mosman Park (1 ML/d)
- supply process and cooling water to industries in Kwinana (initially 15 ML/d)
- a number of tree plantations irrigated with effluent

Treated wastewater is reused on large wastewater treatment plant sites for a variety of purposes, including; chlorine solution water, blower cooling, filter belts washing biogas compression and various sprays. Ongoing expansion of use of effluent is reviewed regularly to reduce the use of potable water.

For the large Perth WWTPs onsite effluent reuse quantities are equivalent to 22 000 households' greywater reuse potential (or 66 000 people). There is potential to double this in the short term.

Irrigation of parks and gardens with treated wastewater has only grown marginally in the last decade but is set to increase significantly as new wastewater treatment schemes are built as part of new schemes or the Infill Sewerage Program. Most of the existing systems suffer from algae laden effluent which causes odours and blockages. Upgrading of the effluent quality will be necessary over the next decade to overcome these problems and ensure effluent reuse continues.

Watering tree plantations with effluent is considered a reuse practise, as an economic return can be realised from the scale of the timber. The largest scheme utilising tree plantations is at Albany in the South-West of Western Australia where an average 6 ML/d of treated wastewater is applied to Tasmanian Blue-gum trees. This is equivalent to 25 000 households (or 70 000 people) utilising their household greywater. In addition a valuable product is available to offset the total cost of treatment and reuse of the wastewater. The whole scheme is operated by two officers who ensure all the public health and environmental safeguards are maintained at the required level.

Several more effluent watered tree plantations are being seriously considered for a number of sites in the South-West of Western Australia. The community is very receptive to this method of utilising effluent and no doubt will lobby strongly to increase the number of these schemes.

In high population areas where groundwater extraction near coastal areas is excessive saltwater intrusion can occur. This is the situation in Mosman Park, a coastal suburb of Perth. Alternative sources of water have had to be investigated to replace the deteriorating quality of the local groundwater that has been used for parkland and golf course irrigation. A feasibility study for a 1 ML/d supply of treated wastewater was undertaken and an amortised unit cost of $2/m^3 was derived. At this stage, the high cost has discouraged scheme proponents to proceed but in due course it is more than likely this decision will be reversed. It is possible that new technology could substantially reduce the cost.

The Kwinana Industrial Area, south-west of Perth has a number of industries that are experiencing difficulties in acquiring sufficient quantities of process/cooling water. A treated wastewater pipeline traverses the industrial area and a feasibility study examining practicalities and costs has been completed.

Approximately 15 ML/d of treated wastewater would be used initially. This project offers a tremendous opportunity to reuse a large quantity of effluent on a daily basis. Delicate negotiations are underway to finalise suitable pricing arrangements for the effluent. Further growth in demand is expected once the project is commissioned.

There are a number of benefits of large effluent reuse schemes compared to numerous small schemes (e.g. greywater recycling) and these include :

- public health and environmental risks are lower
- public health and environmental issues easier to manage
- new technology easier to implement
- new methodologies easier to implement
- economy of size
- lower manning
- regular monitoring and reporting of performance
- easier to expand/change

OTHER CONSIDERATIONS

Whenever a greywater recycling system is installed allowance needs to be made to be able to divert the greywater to sewer or the septic tank system during wet weather or other difficulties. Unless this is provided public health and environmental pollution is at risk. The former is particularly relevant as the most exposed individual, the "toddler" (young child) is present in many households. If an alternative system has to be provided then there are extra costs involved in installing greywater recycling schemes.

One of the big concerns with intermittent/occasional use of the wastewater conveyance and treatment systems is the negative impacts that will arise. If sewers designed for a particular volume of flow (i.e. self cleansing velocity) and treatment plants are designed for particular strength wastewater, then diversions of greywater will have a detrimental impact on the systems. More regular pipe flushing and/or cleaning is likely to be required and blockages will be more prevalent, along with higher odour and corrosion levels. At the treatment plant the quality of wastewater could be twice the strength, and hence impact on odours and effectiveness of operation. Overall there could be extra operation and maintenance costs of up to 20% to cater for "possible" flows of greywater.

CONCLUSION

As a result of several decades of lower than average rainfall and the resistance by the community to development of more large water storages there is a drive to adopt a "water-wise" approach.

As greywater is a large proportion of wastewater produced in households a section of the community and water industry consider that prudent onsite greywater recycling can reduce overall water demand.

Other groups hold the view that there are greater benefits for the community if treated wastewater from WWTPs is used for irrigation, groundwater recharge and industrial purposes.

The effluent reuse option does engender higher community confidence in the protection of public health and the environment, particularly surface and ground waters. The larger schemes offer the further benefits of being more flexible and adaptable to changes, having regular monitoring and reporting and being of lower cost overall.

In more arid areas where deep sewerage is unlikely, properly designed and operated greywater recycling schemes can provide a valuable source of irrigation water.

WETLAND SYSTEMS

A COMPARISON BETWEEN TWO MACROPHYTES ON THEIR NUTRIENT STRIPPING ABILITY OF DOMESTIC GREYWATER

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ABSTRACT

In experiments to determine the effectiveness of particular indigenous wetland macrophyte species to remove nutrients from domestic greywater, a comparison was made between the activity of the root zone and complete pond systems.

One series of experiments focussed on the ability of two plants, *Triglochin huegelii* and *Schoenoplectus validus*, both growing in tanks with water to a depth of 400 mm, to reduce common greywater contaminants such as nitrates, phosphates, ammonium and Biochemical Oxygen Demand (BOD).

In another set of experiments the water was drained from the tanks until it was at a level just below the soil surface. These same plants were then subjected to similar tests, but with reduced greywater input. Only the activity of the root zone, including any uptake or involvement by tubers and rhizomes, was examined.

The uptake of some nutrients, such as nitrate, varies in both species type and plant part. For example, in a pond, the leaves of *Triglochin* clearly absorb nitrate directly from the water. In contrast, the root zone surrounding *Schoenoplectus* seems to be a better absorber of nitrate and ammonium ions.

Ammonium removal was more pronounced. *Schoenoplectus* recorded 54% and *Triglochin* 20% greater removal than the control tanks in the root zone experiments. However, in a pond situation, both *Schoenoplectus,* and *Schoenoplectus* and *Triglochin* together, were greater than *Triglochin* by itself and the control tanks.

BOD reduction also varies from pond to root zone studies. *Schoenoplectus* reduces BOD more than *Triglochin* in root zone analysis, but they are much the same in ponds.

KEY WORDS

Greywater, macrophytes, Schoenoplectus, Triglochin, nitrification, denitrification, volatilisation, root zone, vertical flow.

INTRODUCTION

Most households in Australia produce considerable amounts of wastewater. Dealing with the large volumes of sewage discharged by humans is increasingly becoming a major problem for Government and private wastewater treatment organisations.

Effective on-site sewage treatment for both small and large communities is generally embryonic in Australia. The use of reed beds, stabilisation ponds, soil filters and other types of simple technology is being investigated throughout the world, including Australia, as possible sustainable solutions for our growing wastewater problems. In highly sensitive water catchment areas it is especially important to reduce the levels of nutrients in wastewaters to such a level that no threat to existing water supplies occurs.

Reeds and rushes are useful plants for wastewater treatment. They take up and store nutrients themselves, provide a growing area for micro-organisms, stimulate the soil activity by root excretions and reduce the volume of effluent by transpiration. Thus, aquatic plants offer a technically simple, low cost, energy-efficient method of treating greywater.

Aquatic plant systems require little technical back-up and are easy to maintain. However, what does differ is their individual rate of assimilation and optimal storage level. Brix (1997) has found that the uptake capacity of emergent macrophytes is in the range 30 to 150 $kgPha^{-1}yr^{-1}$ and 200 to 2500 $kgNha^{-1}yr^{-1}$, whereas submergents are generally much lower in the order of less than 100 $kgPha^{-1}yr^{-1}$ and 700 $kgNha^{-1}yr^{-1}$.

The nutrient levels in greywater varies considerably. The quality of greywater effluent primarily depends on the duration of the biological and chemical treatment of the wastewater or upon the detention time (usually in days) as the greywater is undergoing treatment. Treatment of wastewater depends on factors such as system design, the chemistry of the plant root-water-sediment environment, plant uptake, available carbon (for microbe activity), nutrient volatilisation (e.g. ammonia) and type of substrate.

A study of the effectiveness of particular indigenous wetland macrophyte species to remove nutrients from domestic greywater, in both pond and subsurface flow conditions was made. Detailed measurements of BOD, nitrate, phosphate and ammonium concentrations were undertaken. In essence, the comparison was between surface flow and sub-surface flow systems. The aims of the study were to:

- determine the effectiveness of both the root zones and leaf systems (in a pond situation) of wetland macrophyte species to reduce BOD, and levels of nitrate, ammonium and phosphate from known levels in domestic greywater.
- compare the results of the submergent *Triglochin huegelii* (Water Ribbons) with the emergent *Schoenoplectus validus* (Lake Club Rush). *Triglochin huegelii* exists as a small singular clump of primarily vegetative shoots of large fleshy, flattened leaves. *Schoenoplectus* is a larger clumping plant, with stems to 2 m.

MATERIALS AND METHODS

Twelve identical 200 L tanks were set up with four different tank plantings, each of which was repeated three times. Tanks usually contained either four *Triglochin*, two *Schoenoplectus*, or one *Schoenoplectus* and two *Triglochin*. The tanks contained 300 mm of 20 mm stone, 100 mm washed sand and 400 mm water, with one of the following:

1. *Triglochin huegelii* only (Designated "T" in the tables and figures).
2. *Schoenoplectus validus* only (Designated "S" in the tables and figures).
3. *Triglochin huegelii* and *Schoenoplectus validus* combined ("T/S" in tables and figures).
4. No plants - as the control ("C" in tables and figures). These tanks allowed a comparison with the planted tanks, such that the effect of the plants on nutrient levels could be ascertained.

Domestic greywater was collected and stored for five days in a large galvanised rain water tank. A submersible pump mixed the total volume of greywater before samples were taken.

One experiment was conducted as a pond using upward vertical flow to pass fifty litres of greywater through the tank system. Due to the position of taps and outlets in each tank and the frequent cycling of the water through the soil, only downward vertical flow could be used for the below-ground investigation.

In this experiment, only the substrate, consisting of sand and stone as described earlier, was examined and a smaller volume (20 to 40 L) was added to each tank. The water level was kept just below the soil surface. All water was drained from the bottom before refilling with new greywater.

Only the activity of the root zone, including any uptake or involvement by tubers or rhizomes, was examined.

The general method followed for this study was:

1. An initial analysis of the greywater was undertaken for BOD (standard APHA test, 1989). A selection of other parameters, such as nitrate, ammonium and phosphate levels was also measured (using methods outlined in HACH Spectrophotometer manual).
2. Every five days or so, the tanks were either flushed or individually drained and pumped out into another holding container. A one litre heterogeneous sample was taken from this holding or collection tank and this was used for all tests and analysis.
3. This procedure was repeated six times over the course of four weeks for both pond and sub-surface flow studies.

Total nitrogen and total phosphorus, usually determined after digestion, were not undertaken in this study as greywater can be assumed to contain little organic matter, and any nutrients in this form would probably be degraded into inorganic forms which are easier to measure. An earlier study showed that HACH measurements for nitrate and phosphate were about 60 - 80% of Total N and Total P in a number of greywater samples.

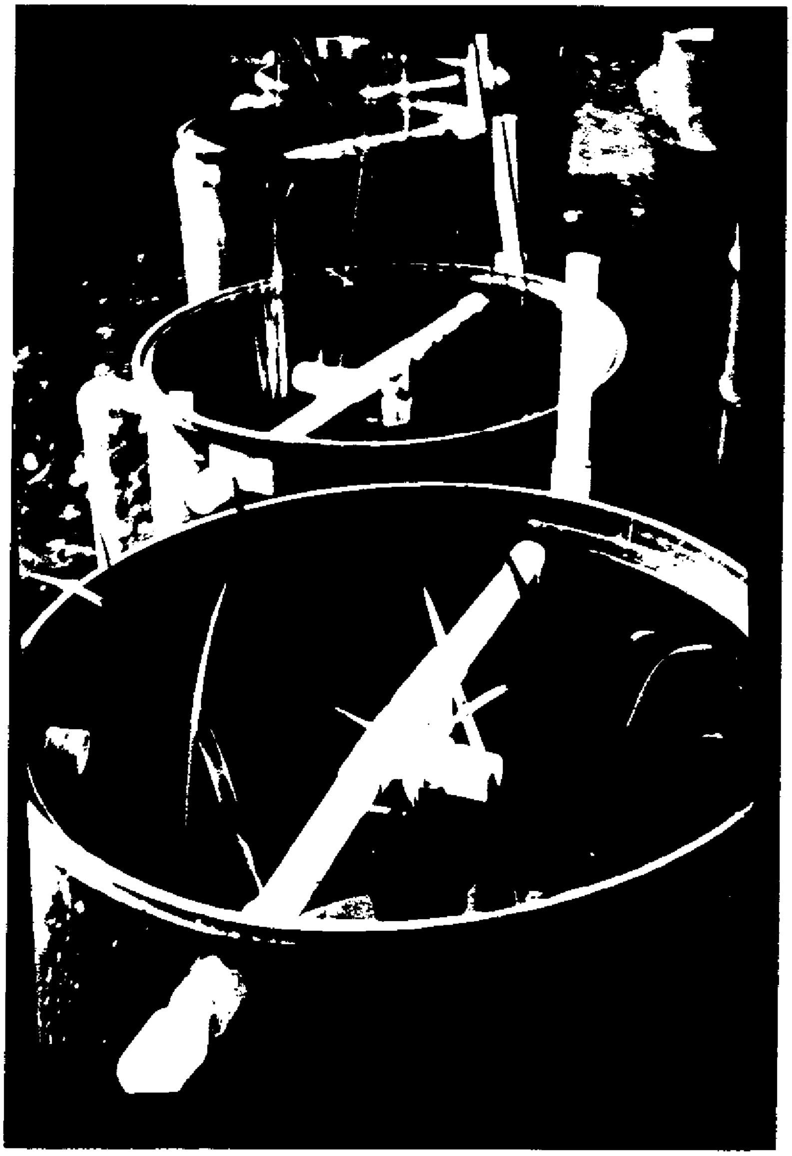

Figure 1 The tank layout for the series of experiments.

Biochemical Oxygen Demand (BOD)

BOD was determined using a HACH 2137 manometric apparatus and 500 mL BOD bottles. A water sample of 157 mL was added to the standard BOD bottle. Two potassium hydroxide pellets were placed in the rubber cup and the bottles were left to mix, without barometric tubes connected, for half an hour. Each manometer mercury level was adjusted to zero after re-connecting the tubes. Bottles were kept at a constant temperature of 20°C as described in the HACH Manual.

Checks were made over the course of five days to make sure that the equipment was functioning properly and thorough mixing was taking place. The final barometric readings were taken after the five day period and are expressed as mg/L BOD.

Nitrate concentration

Nitrate concentration was determined by the cadmium reduction method, using powder pillows with a HACH 2000 spectrophotometer. The method is outlined in the HACH User's Manual. The results were expressed as nitrate-nitrogen in mg/L. If initial testing yielded over-range data, and thus particular nutrient levels were too high, samples were diluted x10 before further analysis.

Phosphate concentration

Phosphate, as orthophosphate, was determined by the ascorbic acid method using powder pillows with the HACH spectrophotometer. This method was adapted from the *Standard Methods for the Examination of Water and Wastewaters*, and was suitable for reactive phosphorus with a concentration of zero to 2.5 mg/L phosphate ions. A description of the method is found in the HACH Manual.

Ammonium concentration

Ammonium was determined by using Nessler's Reagent and the HACH 2000 spectrophotometer. A simple colorimetric method was used to ascertain ammonium ion levels. The method is described in the HACH Manual.

RESULTS AND DISCUSSION

Biochemical Oxygen Demand (BOD)

The reed bed filter system of the tanks provides numerous sites for bacterial attachment, and the extensive root system of both of these types of plants (and thus large surface area of these roots), means that bacterial action will greatly contribute to BOD reduction. Even the stone and soil in the substrate provide these bacterial sites. Table 1 compares the substrate to the pond system and lists the percentage of BOD reduced by each group of tanks.

System	Control tanks	Triglochin tanks	Schoenoplectus tanks	Triglochin and Schoenoplectus tanks
Substrate	83 ± 14	88 ± 13	93 ± 12	92 ± 10
Pond	90 ± 6	95 ± 5	92 ± 6	94 ± 3

Table 1 Percentage reduction in BOD by each system. Standard deviations are shown.
Standard error of substrate means = 4. Standard error of pond means = 2.

BOD is generally greatly reduced in all tanks, with the range of reduction being 83 to 93%. All planted tanks have, on average, greater reduction in BOD than the control tanks, with *Schoenoplectus* higher than *Triglochin* in the sub-surface flow system. The results for *Schoenoplectus* and the combined planted tanks are statistically significant. In the pond system, *Triglochin* has slightly greater reduction in BOD than *Schoenoplectus*, with combined planted tanks in between. Only the *Triglochin* results can be said to be statistically significant.

Figure 2 shows the general relationship between the BOD loading rate and the amount of BOD removal. It seems that the amount of BOD removed is proportional to the loading rate, except at high loading rates where it decreases. At higher rates, a longer retention time may be needed by these types of plants to reduce BOD.

A typical analysis of BOD removal is shown in Figure 3 (next page). Here, the amount of BOD input (loading) and then removal for each set-up is shown, with planted tanks consistently removing more BOD than unplanted tanks.

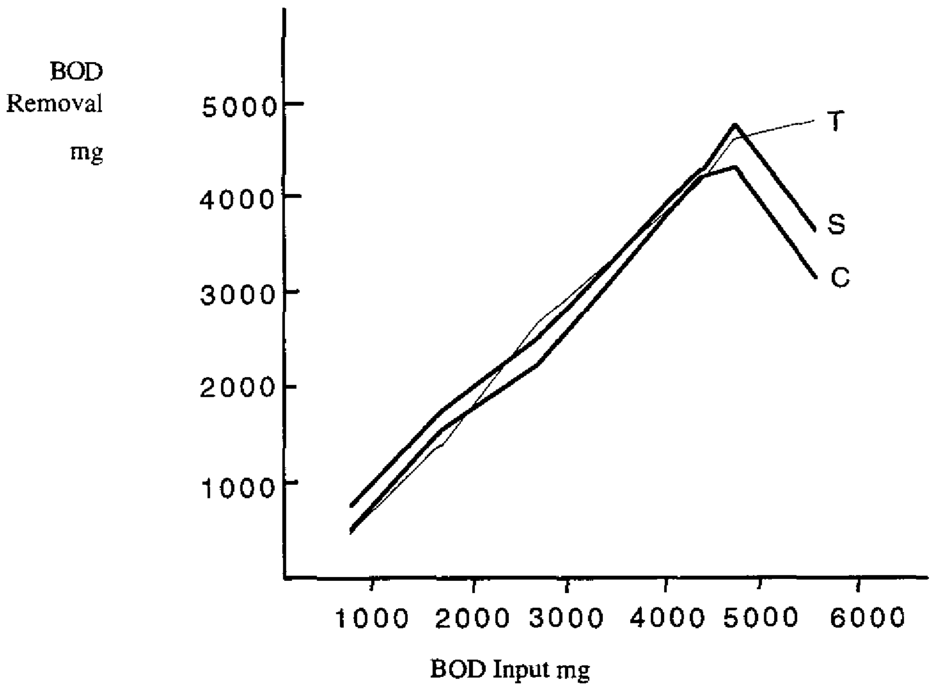

Figure 2 The effect of BOD loading on BOD removal.
In all such figures, C = control, T = *Triglochin* and S = *Schoenoplectus*.

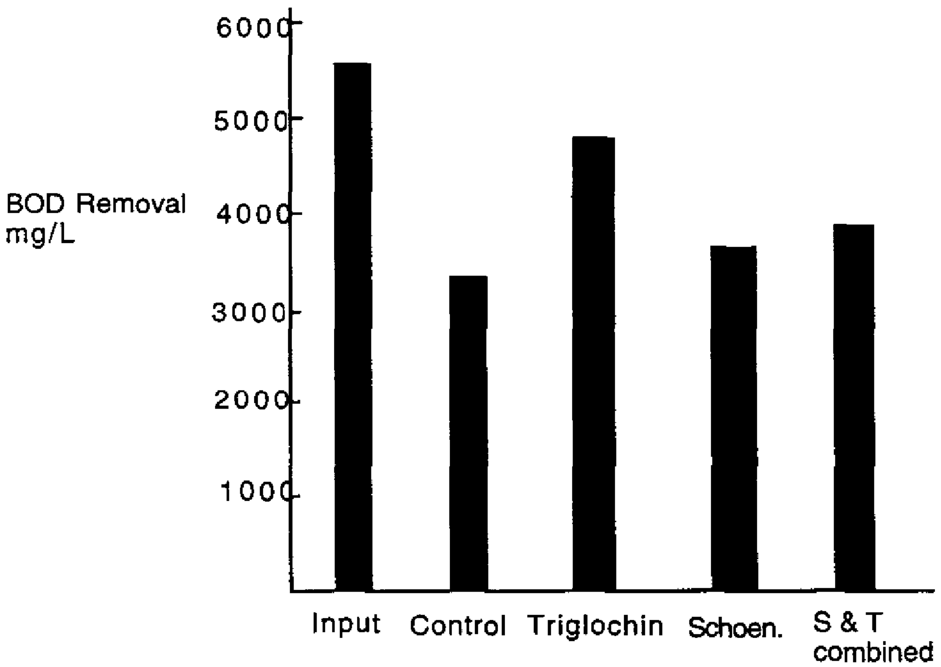

Figure 3 BOD removal for Trial 1 substrate - total loading = 5680 mg. Standard deviations are shown.

Nitrate concentration

Kadlec and Knight (1996) state that the factors affecting the total nitrate removal rate in a natural wetland include total nitrogen loading rate, climate, plant community composition and soil characteristics.

Total nitrogen mass removal rate depends on the forms of nitrogen in the inflow, water depth, dissolved oxygen and total nitrogen mass loading rate. The absolute total nitrogen mass removal rate has been found to correlate consistently with total nitrogen mass loading rate up to loading rates of 30 $kgha^{-1}d^{-1}$ (3 $gm^{-2}d^{-1}$). At higher loading rates, mass removal efficiency declines.

This investigation had similar results. The effect of macrophytes in nitrate removal can also be seen in Figures 4 and 5. Figure 4 shows that relationship between nitrogen loading and the amount of nitrate-nitrogen reduction by the tanks. There seems to be a direct, almost linear, relationship between the two, at these low loading rates.

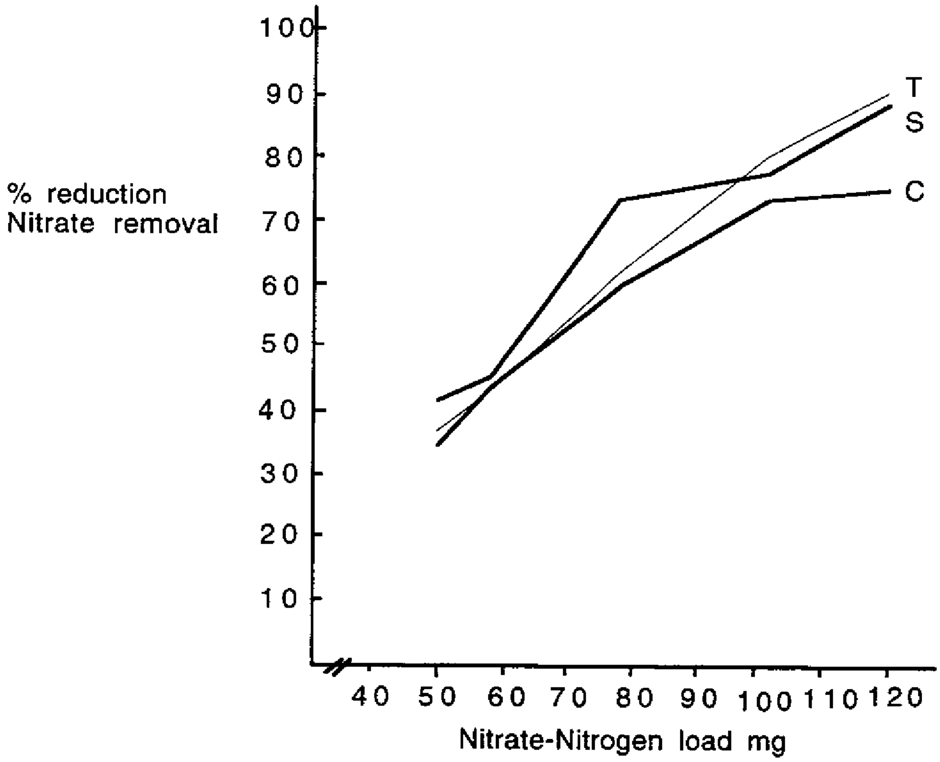

Figure 4 The effect of nitrate-nitrogen loading on nitrate removal.

From Table 2, all planted tanks, except *Schoenoplectus* in the pond, have, on average, more nitrate removal and reduction than the unplanted control tanks. In a pond, *Triglochin* had greater nitrate removal (by 3%) than the control tanks, which were themselves better than *Schoenoplectus* tanks. When only considering root zones, *Schoenoplectus* was 8% and *Triglochin* 4% greater than the control tanks. However, only some of these results are statistically significant. For example, the *Triglochin* and *Schoenoplectus* substrate-only tanks are statistically significant from the unplanted control tanks, but the *Triglochin* and *Schoenoplectus* pond tanks were not.

Again, as previous experiments by Mars *et al.* (1996) have indicated, the combination of *Triglochin* and *Schoenoplectus* seem to cause lower assimilation of some nutrient parameters. There may be some antagonism between the plants, for reasons unknown at this stage of our investigation with these plants.

In a pond system, *Triglochin* had a greater uptake of nitrate than *Schoenoplectus* even though the results are not statistically significant. This does, however, support an earlier experiment (Mars *et al.*, 1996) where similar results were obtained. It appears that *Triglochin* may be able to absorb nitrate more efficiently through its leaf system than through its roots.

System	Control tanks	Triglochin tanks	Schoenoplectus tanks	Triglochin and Schoenoplectus tanks
Substrate	71 ± 5	75 ± 3	79 ± 6	73 ± 7
Pond	70 ± 6	73 ± 4	64 ± 10	71 ± 7

Table 2 Percentage reduction of nitrate by each system. Standard deviations are shown.
Standard error of substrate means = 2. Standard error of pond means = 2.

To examine the general trend of planted and unplanted tanks in their ability to reduce nitrates in the greywater, the results of one typical trial are shown in Figure 5. Again, the differences between the various plants can be easily seen, with the results for *Schoenoplectus* statistically significant.

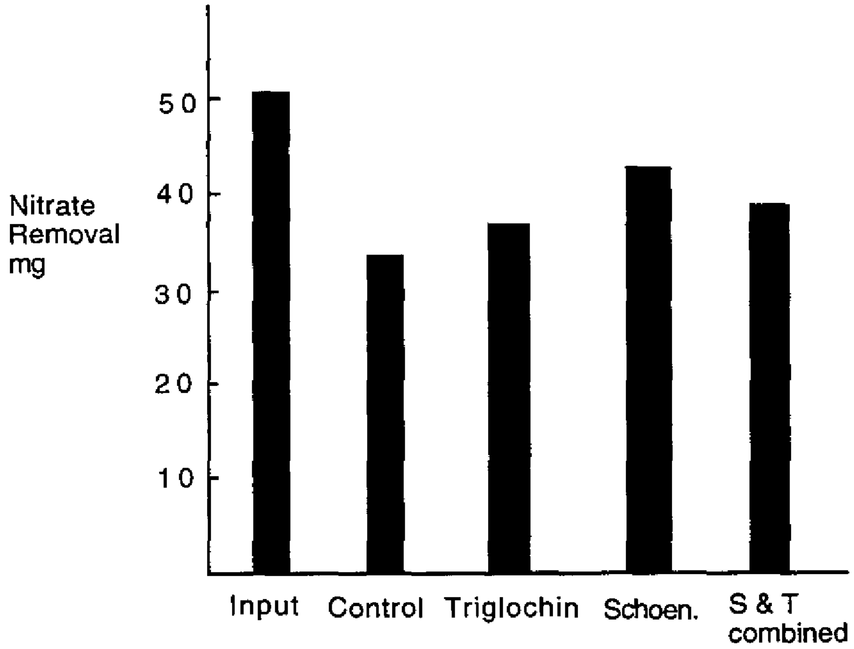

Figure 5 Nitrate removal for Trial 6 - total nitrate-nitrogen loading = 50 mg. Standard deviations are shown.

Phosphate concentration

Large amounts of phosphorus are stored in the substrate, bound to soil particles and organic matter. This is possibly the reason for high phosphate retention in all tanks, including the control tanks which did not contain plants.

System	Control tanks	Triglochin tanks	Schoenoplectus tanks	Triglochin and Schoenoplectus tanks
Substrate	99.0 ± 0.5	99.6 ± 0.1	99.7 ± 0.2	99.6 ± 0.4
Pond	88 ± 3	71 ± 17	86 ± 1	91 ± 4

Table 3 Percentage reduction of phosphate by each system. Standard deviations are shown. Standard error of substrate means = 1. Standard error of pond means = 1.

All tanks in the substrate-only experiment had exceptionally high phosphate retention. In comparison, the pond system study had much less phosphate assimilated - possibly due to some orthophosphate in solution passing out of the tanks. None of the results are statistically significant and no plot of phosphorus has been included here.

It appears that plants do not assimilate or absorb much phosphate - rather it is absorbed onto substrate particles. Other studies (Mars, 1997, unpublished data) have shown that the levels of phosphate in many wetland species fluctuate and varies from plant to plant and within plant organs.

Ammonium concentration

Ammonium-nitrogen is both produced and consumed in wetlands. Organic nitrogen is converted to ammonia, and ammonia is changed into nitrates and nitrogen gas. Ammonia volatilisation is reduced in subsurface flow but not in open water surface flow systems. Ammonia volatilisation increases with mixing, but it is not the main way ammonia is removed. Ammonia (gas) is generally only a small fraction of the total ammonium, which exists as an ion in solution. Green *et al.* (1997) found that nitrification was the major mechanism for ammonia removal while other mechanisms (such as ammonia volatilisation) were negligible.

Kadlec and Knight (1996) quote that the fraction of un-ionised ammonia (NH_3) is only 0.6% at 25°C and pH 7 and this increases to 70% at a higher pH and temperature (pH 9 and temperature of 30°C). A slowly increasing pH level was noted in some tanks in the pond experiment, and this may account for some ammonia removal.

Von Felde and Kurst (1997) found that both COD and ammonium degradation decreased when their columns were flooded or flushed frequently - possibly there was less oxygen available in the water (oxygen diffuses ten thousand times slower in water than in air). This is a good case for resting the beds or draining all water and replacing them with air for one day or so before refilling with wastewater.

The loss of ammonium ions from the system can be high as some of the ammonium ions may be changed to ammonia gas, which can be lost to the atmosphere by ammonia volatilisation. Some ammonium ions will also be oxidised to nitrates by nitrifying bacteria. Plants probably do not take up ammonium as itself, although some can be adsorbed onto plant tissue - it has to be changed to nitrate first.

For these reasons, it is not surprising to suggest that much of the ammonium is lost from the aquatic system rather than being absorbed and then utilised by plant tissue. This idea can be seen in Figure 6 where the ammonium removal rates were generally very high.

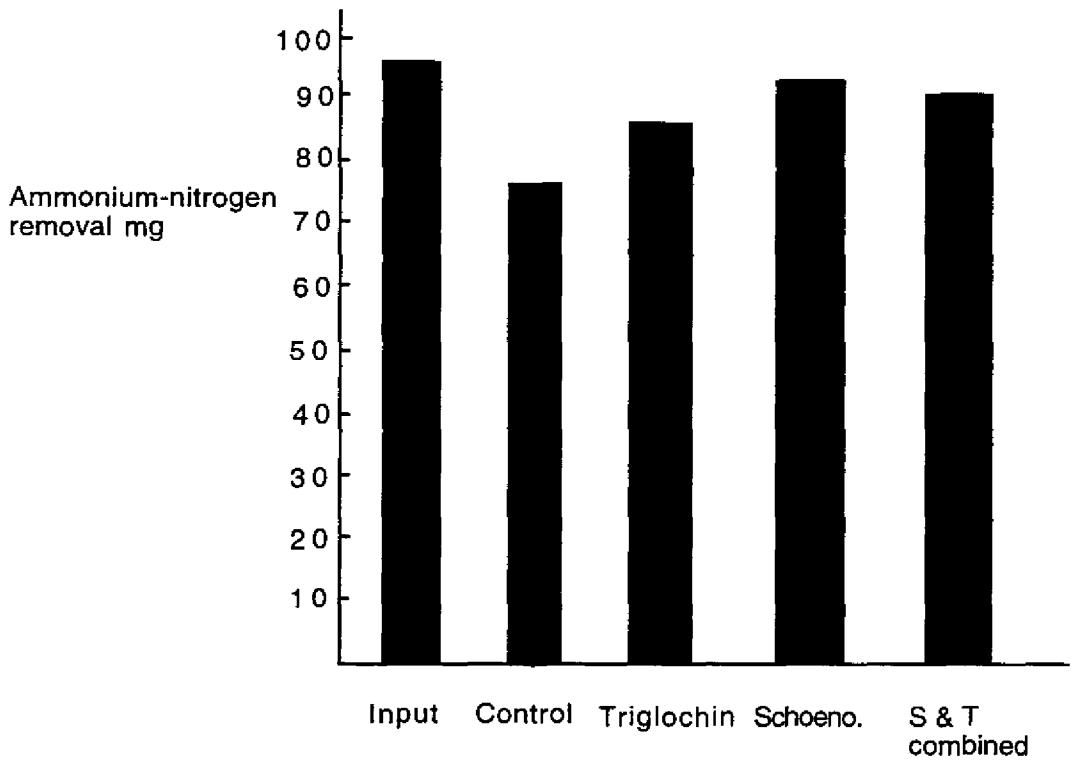

Figure 6 Ammonium-nitrogen removal for Trial 2 - total loading = 96 mg. Standard deviations are shown.

In both studies, there is significant difference between the planted tanks and the control tanks. *Schoenoplectus* and the combined plant tanks have a significantly greater utilisation or effect on ammonium concentration. It could be that these plants provide suitable conditions for bacterial action around the root zone.

In the sub-surface flow study, ammonium reduction in *Schoenoplectus* was 54% greater than that of the control tanks. Even *Triglochin* was 20% better than the control tanks, whereas in the pond system it was much the same. All of these results, except *Triglochin* in a pond, are statistically significant, with the differences between means usually more than three times the standard error of the differences between the control tanks and the planted tanks.

System	Control tanks	Triglochin tanks	Schoenoplectus tanks	Triglochin and Schoenoplectus tanks
Substrate	64 ± 16	79 ± 10.5	99 ± 1	95 ± 2
Pond	86 ± 8.5	88 ± 6.5	96 ± 3	95 ± 3.5

Table 4 Percentage reduction of ammonium by each system. Standard deviations are shown. Standard error of substrate means = 2. Standard error of pond means = 2.

The wide range of results from each of the trials could be due to the daily temperature variation that occurred throughout the experiment. Changing pH and water temperature may affect the volatilisation of ammonia and activity of microbes. The processes of ammonification, nitrification and denitrification are all temperature dependent. Even so, the pond system does seem to provide suitable conditions for ammonia volatilisation or loss.

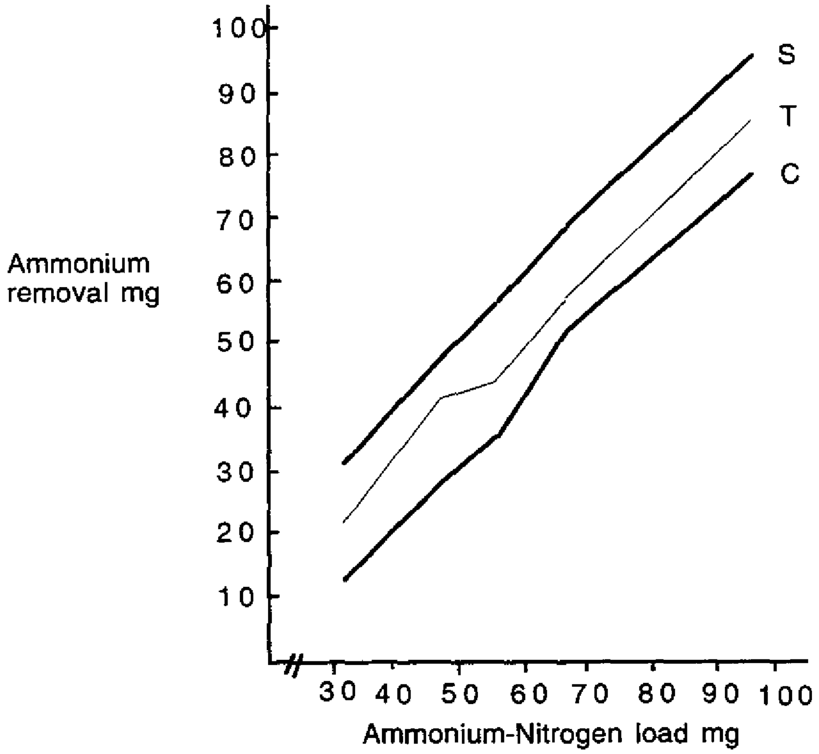

Figure 7 The effect of ammonium-nitrogen loading on ammonium removal.

At low loading rates, there is a direct correlation with ammonium removal. Figure 7 clearly shows the ability of *Schoenoplectus* and *Triglochin* to remove ammonium from the wastewater.

CONCLUSIONS

The treatment and disposal of greywater can be accomplished simply by mimicking the cycling of matter in nature. Wetland plants are well known as fast-growing accumulators of nutrients. For example, our studies have shown that planted tanks, in both the pond and substrate systems, have, on average, more nitrate removal and reduction than the unplanted control tanks.

In a pond system, *Triglochin* had greater uptake of nitrate than *Schoenoplectus*, as it appears that *Triglochin* may be able to absorb nitrate more efficiently through its leaf system than through its roots.

Removal of ammonium is similar. In both studies, there is significant difference between some of the planted tanks and the control tanks. *Schoenoplectus* and the combined plant tanks have a significantly greater utilisation or effect on ammonium concentration. *Triglochin* was marginally better than the control tanks, and more experiments may reveal the reasons for this discrepancy.

It must be kept in mind that the emergent *Schoenoplectus* is a faster growing clumping plant, whereas the submergent *Triglochin* is a singular plant. It is fair to say that the *Schoenoplectus* tanks had far greater biomass present, and future studies will need to consider the amount of nutrient reduction compared to plant biomass.

Initial studies (Mars *et al.*, 1996) have already compared nutrient uptake by leaves, roots, tubers as dry mass, and results have shown that these types of plants do effectively assimilate nutrients from greywater. Unless these plants are harvested much of this stored nutrient is recycled in the waterway as plants die and decompose. Long term storage of nutrients best occurs when they are precipitated and deposited as particulate chemical compounds.

This is how phosphate is probably removed from wastewater. Even so, there is a difference in the amount of phosphate removed in each system. All tanks in the substrate-only experiment had exceptionally high phosphate retention, while the pond system study had much less phosphate assimilated. Further studies will be required to determine the reasons for this.

Finally, BOD is generally greatly reduced in all tanks, with planted tanks having, on average, greater reduction in BOD than the control tanks. *Schoenoplectus* had higher BOD removal than *Triglochin* in the sub-surface flow system. In the pond system, *Triglochin* has slightly greater reduction in BOD than *Schoenoplectus,* with combined planted tanks in between.

These experiments have also shown the general relationship between the BOD loading rate and the amount of BOD removed. BOD removal is proportional to the loading rate, except at high loading rates where it decreases. At higher rates, a longer retention time may be needed by these types of plants to reduce BOD. Longer retention times will also, no doubt, increase the removal efficiency of other nutrients such as nitrate, ammonium and phosphate ions.

FUTURE CHALLENGES

At present, reed beds, which are designed for wastewater treatment, are mainly used in tertiary treatment of domestic wastewater and for polishing treated effluent from industry. Knight (1997) adds that treatment wetlands should be considered holistically as they can also provide wildlife habitats, educational benefits, and facilities for general public use.

In the near future, responsible living practises will likely include the separation of blackwater from greywater and, as Fittschen and Niemczynowicz (1997) and Henze (1997) have suggested, even the separation of urine and/or kitchen wastes from other types of greywater (from bathrooms and laundry). Studies such as these contribute to our understanding of the complexities which occur in a wetland ecosystem, and to finding sustainable ways to treat our wastes.

Researchers should focus their energy and work to produce wetlands that not only treat and purify wastewater, but produce plant products, which can be used for fodder, paper production, thatching and fencing, as well as animals such as crustaceans and fish which can be periodically harvested as food. Too often, we focus on only treating wastewater without the vision of an integrated sustainable polyculture which provides many more benefits to humans, animals and the environment.

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PERFORMANCE OF A CONSTRUCTED WETLAND RECEIVING HIGH HYDRAULIC AND CONSTITUENT LOADS

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ABSTRACT

Performance data for constructed wetlands suggest that they have an important role to play in polishing domestic wastewaters prior to their disposal. Agricultural wastes however, with their often high hydraulic and constituent loads, offer particular challenges to the pollutant removal efficiencies of wetlands.

In 1995 a surface flow constructed wetland was built to treat dairy wastewaters at Tocal Agricultural College in the Hunter Valley, N.S.W. The system was designed for a herd size of 110 animals with a detention time of between 10 and 14 days. Three native species of macrophytes were planted in the wetland, while effluent pre-treatment was undertaken using a gross solids separator and anaerobic pond. Effluent volumes and quality entering and leaving the wetland system were monitored.

This paper describes some of the preliminary results achieved by the wetland with respect to the reduction of organic matter and nutrients. Hydraulic loading to the wetland was higher than anticipated during the study and as a result the pollutant removal efficiencies were variable. Reductions in BOD were much higher than those for Total Nitrogen and Total Phosphorus. While the high concentrations of organic matter and nutrients in the dairy effluent were reduced through the constructed wetland, further treatment or land application of the effluent is required.

KEYWORDS

Dairy waste; macrophytes; BOD; nitrogen; phosphorus;

INTRODUCTION

Research which has been conducted into the performance of constructed wetlands for the treatment, and particularly the polishing, of domestic wastewaters has led to the construction of many municipal treatment wetlands. In general, reported pollutant removal efficiencies are often high for a number of parameters, but wetland performance is variable due to the individual plant species used, vegetation die-off or senescence, variations in hydraulic loads and water levels, and other difficulties which are inherent in managing natural treatment systems (Kadlec & Knight, 1996).

In recent years, efforts have been made to assess the suitability of constructed wetlands for the treatment of agricultural wastewaters which typically have significantly higher concentrations of organic matter and nutrients (DuBowy & Reaves, 1994). As these high concentrations can lead to water management problems if allowed to discharge directly to receiving waters, it is essential that these wastewaters be treated on-farm using stabilisation ponds, or constructed wetlands, provided they can accommodate the high hydraulic and pollutant loads which are often generated.

Within Australia and elsewhere, there is now a requirement for upgrading waste management systems in intensively managed animal agriculture. Research which has examined the performance of constructed wetland systems for the treatment of dairy wastewaters has shown that significant improvements in effluent quality may be achieved (Tanner, 1994; Moore et al., 1995; Cronk, 1995) due to the physical, chemical and biological processes which occur in wetland systems. While there are some difficulties in directly comparing pollutant removal efficiencies because different studies use different reporting formats, constructed wetlands do show considerable potential for reducing biochemical oxygen demand (BOD), suspended solids, faecal coliforms and nutrients. In addition, a number of other factors such as the initial concentrations of pollutants, the hydraulic load generated, the type of pre-treatment used and the design characteristics of the wetland may all affect the reported performance of a dairy wastewater treatment wetland.

Best management practice for the Australian dairy industry has been described by Wrigley (1994) and may include collection of milking shed wastes, pre-treatment, waste stabilisation pond(s) and land application by spray irrigation. While there has been research undertaken on a number of aspects of dairy waste treatment and disposal, for example nitrogen leaching from irrigated dairy effluent, little work has been undertaken on the use of constructed wetlands for the treatment of dairy wastewaters, in particular the removal of nutrients, prior to its disposal to land. Masters (1993) described a low cost dairy waste treatment system using phosphorus sorbing materials for a farm in W.A., while Geary et al.,(1995) reported on the usefulness of wetland filter strips in treating dairy effluent which had received pre-treatment in a two-pond storage system.

In 1995 the Dairy Research & Development Corporation and the Hunter Catchment Management Trust funded the construction of a wetland system at Tocal Agricultural College, near Maitland in N.S.W.. The objectives of the project were to examine the pollutant removal efficiencies which could be achieved by a constructed wetland receiving dairy wastewaters and to report on the suitability of such a natural treatment system for dairy waste management.

MATERIALS AND METHODS

The dairy waste management system which existed at Tocal Agricultural College prior to 1995 consisted of a coarse solids separator, a small two-pond (anaerobic/facultative) storage system followed by spray irrigation to land. The modern dairy, which is managed by College staff and utilised in teaching agricultural students, is cleaned daily with water from two sources. Town water is used for the cleaning of milk storage vessels, while river water and a large diameter hose are used to clean the dairy and concrete yards. All water used, apart from roof runoff, enters the dairy waste management system. Milking herd size typically varies between 110 and 130 head.

Prior to commencing the study, water samples from the two-pond storage system were taken and analysed and each pond volume was surveyed. The results indicated that the effluent quality was quite poor and that inadequate on-farm storage existed, particularly during wet weather periods. Existing pond volumes were calculated as 280 m^3 (anaerobic) and 120 m^3 (facultative). Some initial remediation work such as the redesign and enlarging of the coarse solids separator and sludge removal from the anaerobic pond were undertaken with immediate improvements in wastewater quality, but the system was still considered to be hydraulically overloaded.

In late 1995 two 32 m long surface flow wetland trenches were constructed, shown schematically in Figure 1. Wetlands 1 and 2 (W1 & W2) had surface areas of 127 m^2 (average depth 0.2 m) and 168 m^2 (average depth 0.5 m) respectively. In the design of the wetlands, estimates of water use were derived after consultations with dairy staff with the intention that only part of the wastewater stream would flow by gravity through the wetlands. However, in the end, all dairy wastewaters were directed into the wetlands which resulted in theoretical detention times in W1 and W2 of approximately four and ten days respectively.

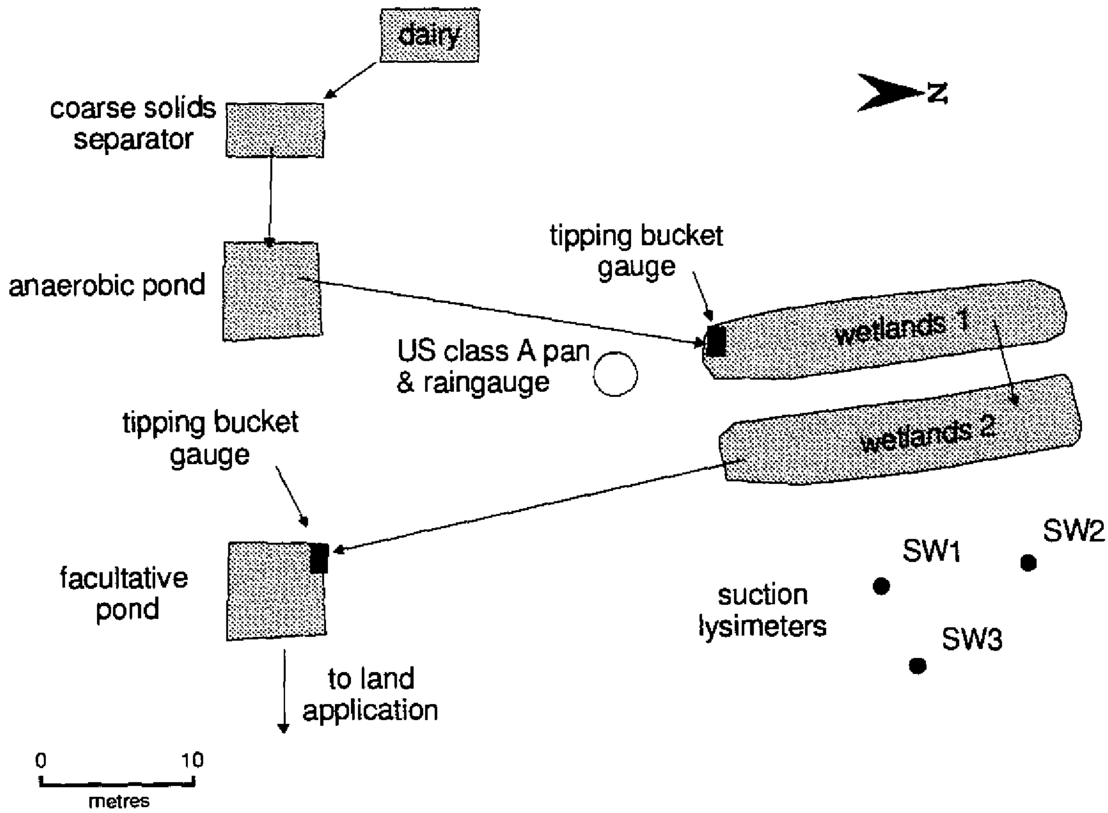

Figure 1 Tocal Agricultural College Dairy Effluent Management System

After the trenches were constructed, an impermeable, synthetic liner was positioned on the base and downslope walls of each wetland. A 0.2 m layer of bentonite clay was then placed on top of the liner and overlain by 0.3 m of topsoil which was obtained from the excavation. Three species of native macrophytes (*Baumea articulata, Phragmites australis* and *Schoenoplectus mucronatus*) were hand planted in broad species bands (approximately 5/m^2) in October 1995. The plants were then grown over the summer in shallow clean water obtained from the pasture irrigation system. Water was added to the trenches as the plants grew until April 1996, when dairy effluent was diverted into the wetlands.

During 1996 instruments were installed at the dairy and at the wetland to measure a number of hydrological variables. Flow meters were fitted to both water sources used in dairy cleaning operations, and two large tipping bucket gauges (approximately 6 L/tip) were used to measure inflows and outflows from the wetlands (Figure 1). During 1997 a data logger was installed on-site to log flow rates and to verify the number of tips recorded from the mechanical counters on each bucket. A non-recording rain gauge and U.S. Class A evaporation pan were also placed on-site, although longer term daily records are also available from the College's meteorological station, approximately 1000 m away.

Other instruments which were installed included a series of tensiometers and suction lysimeters, particularly downslope from the second wetland, to enable an assessment to be made of the sub-surface loss for water balance calculations. Monitoring of a number of water quality parameters is being undertaken at three locations in the wetland; inlet W1, outlet W1 and outlet W2. The following field analyses are conducted weekly using a Horiba Water Quality Checker: pH, electrical conductivity (EC), dissolved oxygen (DO), temperature and turbidity, while monthly determinations of biochemical oxygen demand (BOD_5), ammonia (NH_3), total Kjeldahl nitrogen (TKN) and total phosphorus (TP) are undertaken in a NATA registered laboratory. Both total oxidised nitrogen (TON) and orthophosphate (total reactive phosphorus) are monitored periodically in the wetland. Monitoring of these parameters commenced in April 1996 and is continuing. A number of other investigations into the evapotranspiration rate of wetland plants, the phosphorus sorption of wetland sediments, hydraulic residence time determination using tracers and soil water chemistry are also underway.

This paper presents some of the initial results related to the dairy wetland water balance and the water quality monitoring program. The water balance research is reviewed in an attempt to establish sub-surface leakage from the constructed wetlands, and the monitoring data analysed to determine the pollutant removal efficiencies achieved.

RESULTS AND DISCUSSION

The results of the monthly inflow/outflow monitoring to the wetland are shown in Table 1. The theoretical residence times and hydraulic loads which are shown are only approximate due to the daily variations in water use and other important variables such as rainfall and evaporation.

Monitoring of flows has sometimes proved difficult because of the significantly higher than anticipated water use within the dairy and the intense rainfall events which occur in this near coastal location. Based on the monthly figues contained in Table 1, theoretical residence times of wastewater in the wetland vary between 10 and 18 days, which appears to be similar to the original design figures. While this design assumed gravity flows with continuous throughflow of effluent, the presence of the coarse solids separator and the anaerobic pond before the wetland have resulted in further variations in the wetland's hydraulic loading. In addition, there have been periodic blockages of effluent, and when released, this has surged through the wetland as plug flow. Analysis of the daily flow record illustrates this point. Between the 19 and 20 June, 1997, following heavy rain and a temporary blockage, approximately 30.6 m^3 entered the wetland, and 29.6 m^3 exited.

The monthly data clearly indicate that less water leaves the wetland than enters. This is not unexpected given evaporation and transpiration from water and plants respectively, and the possibility of loss by infiltration, as most liners are not totally impermeable. Soil water sampling using suction lysimeters downslope from the wetland has indicated that a small component of leakage should be considered in the wetland water balance. The presence of high concentrations of nitrate in several of these samples would suggest subsurface leakage of effluent from the wetland.

TABLE 1. Monthly Inflow/Outflow Volumes and Water Balance Data

Month	Inflow (m^3)	Outflow (m^3)	Hydraulic Load (mm/day)	Residence Time (days)
July	247.4	225.3	27.0	12.5
August	234.0	189.0	25.6	13.2
September	252.6	226.8	28.5	11.9
October	***203.9***	175.7	22.3	15.2
June-96	269.9	256.6	30.5	11.1
November				
December	***236.9***	209.7	25.9	13.1
January-97	***195.4***	166.9	21.4	15.9
February				
March	198.8	***171.7***	21.7	15.6
April	163.9	137.1	18.5	18.3
May	253.7	***225.0***	27.7	12.2
June	240.7	215.8	27.2	12.5
July	287.1	251.1	31.4	10.8
August	185.3	166.1	20.3	16.7
September	232.5	***204.4***	26.3	12.9

Note: Bold italics represent adjusted inflow/outflow volumes where there was an instrument measurement error. Hydraulic load is based on a surface area of 295 m^2, while residence time is calculated using a wetland volume of 100 m^3.

While it is not possible to undertake an analysis of the water balance using monthly data for reasons previously discussed, an attempt has been made to quantify the infiltration component for selected parts of the record when there has been little rainfall and when continuous low throughflow from dairy use has occurred. Short term data for periods of between three and seven days have been examined using the simplified water balance equation below (modified from Linacre,1976),

$$S = P + I + \Delta H - O - EV \qquad (1)$$

where S is seepage into soil beneath, P is rainfall, I is inflow to wetland, ΔH is fall in water level (assumed to be zero under these conditions), O is outflow from wetland and EV is evapotranspiration.

Flows from the wetland, and rainfall and evaporation data from Tocal Agricultural College have been used in the above equation. Evapotranspiration has been estimated as being equal to Class A pan evaporation data using a pan factor of 0.7 (typically used to relate pan results to open water surfaces) and assuming that wetland evaporation is in excess of lake evaporation. A conservative crop factor of 1.4 for macrophytes has then been used (Boyd, 1987) to estimate evapotranspiration. The results of this analysis have been graphed in Figure 2 which relates Class A pan data with wetland loss (evapotranspiration plus infiltration). On this basis, it appears that the majority of the flow reductions through the wetland are due to the evapotranspiration component rather than soil infiltration. The loss factors for infiltration of only several hundred litres/day are quite small in relation to the daily hydraulic loads to the wetland which can vary between 6 and 10 m^3. It has been assumed that the saturated conditions under the wetland will result in constant leakage in further water balance work.

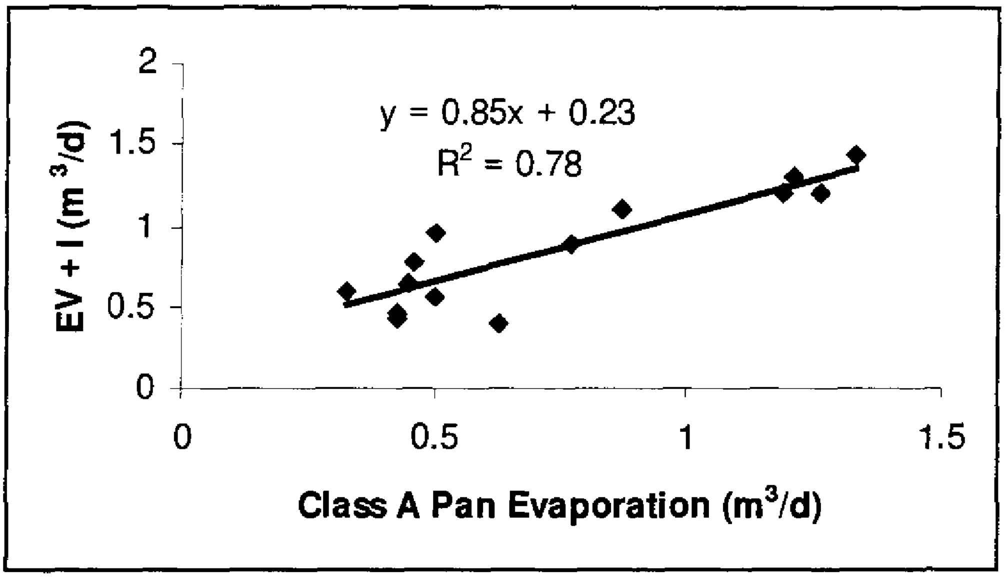

Figure 2 Water Balance Loss Factors

The monitoring data for BOD_5, Total Nitrogen (TN), Total Phosphorus and Total Dissolved Solids (TDS) for the period June, 1996 to September, 1997 have been summarised and are presented in Table 2. The Total Nitrogen concentrations were derived from the sum of the individual nitrogen components (TON+TKN), while the TDS concentrations were calculated from electrical conductivity (uS/cm) multiplied by 0.65 (Hem, 1970).

TABLE 2. Wetland Water Quality (Concentrations in mg/L)

	Mean	Number	Max	Min
BOD_5 -In	168	16	263	64
BOD_5-Out	72	16	180	16
Total N-In	250	16	364	139
Total N-Out	179	16	306	88
Total P-In	59.7	16	79.3	40.2
Total P-Out	45.2	16	68.0	24.2
TDS – In	2320	57	3243	1352
TDS – Out	2036	57	2983	1131

The effluent quality appears to be typical of dairy runoff as it contains high concentrations of organic matter and nutrients, and its quality often varies. While it is apparent that effluent concentrations are improved by passage through the wetland, it is more appropriate to examine the performance of the wetland using a mass balance approach. This approach incorporates hydraulic loading and precipitation (which may dilute effluent quality) and evapotranspiration (which may concentrate). The determination of budgets is important for understanding the nutrient removal functions of wetlands (Kadlec, 1986).

The results of the mass balance calculations are presented as pollutant removal efficiencies in Table 3 for the period of monitoring.

TABLE 3. Calculated Pollutant Removal Efficiencies

Month	% BOD_5	% Tot N	% Tot P	% TDS
June-96	16.5	41.0	35.0	21.3
July	42.3	25.9	34.4	21.6
August	35.6	32.1	43.6	27.6
September	43.7	28.2	25.2	18.4
October	69.7	38.8	43.9	21.8
November				
December	77.1	38.4	32.5	21.3
Jan-97	75.9	29.5	16.0	25.0
February				
March	87.5	31.7	13.6	24.6
April	88.3	53.5	37.9	29.2
May	43.2	31.0	29.8	23.3
June	72.6	41.1	48.8	21.5
July	46.5	33.3	-1.7	24.7
August	57.6	42.8	37.9	24.2
September	38.5	51.6	47.6	23.8

While this input-output budget for the Tocal dairy wetland only summarises the short term net function of the system, it yields little information on the mechanisms by which these results are achieved. Nevertheless, this black-box level of data interpretation suggests that the constructed wetland achieved variable and sometimes high pollutant removal rates over the monitoring period. From hydraulic load and concentration data, the average mass loadings to the wetland using a surface area of 295 m^2 were 5.6 $g/m^2/d$ BOD_5, 5.8 $g/m^2/d$ TN, 1.5 $g/m^2/d$ TP and 46.9 $g/m^2/d$ TDS with average pollutant removals of 57%, 37%, 32% and 23% respectively. These figures are however highly variable as the above table suggests.

Constructed wetland studies have generally reported significant reductions in BOD, due to the additional detention provided by further storage and the presence of plants assisting with sedimentation and filtration. The organic matter reductions are also due to various aerobic/anaerobic decomposition processes within the wetland ecosystem. In relation to the Tocal data, it appears the BOD_5 reductions are significantly higher in the warmer water temperatures over summer (December – April), with reductions of up to 88% being achieved in one month. Wetland 1 does receive some sludge carry-over from the anaerobic pond, however, the wetland design prevents this continuing into Wetland 2 and, as a result, solids are typically reduced in this trench. The high organic loading of the wastewater to Wetland 1 quickly consumes any available oxygen and creates an anaerobic environment, while dissolved oxygen is usually present in low concentrations (<4 mg/L) at the outlet to Wetland 2.

Animal waste is characterised by high concentrations of ammonia and organic nitrogen. The removal efficiencies of Total Nitrogen achieved by the wetland appear to be due to limited nitrification in Wetland 2 and atmospheric loss of ammonia. The wetland is considered to be oxygen-limited at these high loading rates, so there is little opportunity for significant nitrification in the effluent to occur. Oxygen leakage from the roots may however create oxidised conditions in the anoxic substrate and this is considered to stimulate both aerobic decomposition of organic matter and growth of nitrifying bacteria (Barko et al., 1991). Ammonia concentrations are high and typically 50% of the Total Nitrogen and one concern at this high loading rate is toxicity to the growth of plants. While the *Baumea* and *Schoenoplectus* are particularly healthy in the wetland, the *Phragmites* plants have not grown well. This may be due to the deeper water level in Wetland 2 (0.5 m) or the high ammonia concentrations in the effluent, however, it has not been examined further.

Phosphorus removal rates are quite variable, but they are still relatively high given the high loads to the wetland, with up to 50 % of the monthly load being removed during one month. The exception to this is July 1997 when the wetland appeared to release phosphorus. While phosphorus may be immobilised in constructed wetlands by plant uptake, adsorption, precipitation and incorporation into biological films, P removal is highly sensitive to loading rate and considered finite subject to the ability of the substrate to sorb phosphorus. It generally declines after an initial equilibration period, and while a study by Cooke (1992) has shown that high removal rates can be achieved (principally in deposition) in a highly-loaded wetland, it remains to be seen how this wetland will perform in relation to P removal in the longer term. A large portion of this short-term P removal is considered to occur in Wetland 1 in association with the deposition of sludge carried over from the anaerobic pond.

Total dissolved solids represent the sum of the inorganic salts present in the wastewater. High concentrations primarily originate from the animal manure and various cleaning agents which are used in the dairy. While significant removal efficiencies would not be expected for this conservative pollutant, the analysis shows that between 20 and 30% is removed by passage through the wetland. It is suggested that this is primarily as a result of sedimentation processes (from the sludge carry-over and settlement) and minor loss of, for example nutrients, which are utilised by plants.

The results presented here are preliminary only. A final report will be prepared for the Dairy Research & Development Corporation which will address the suitability of wastewater wetland treatment systems for dairy waste management. A similar project at the Ruakura Research Farm in Hamilton (N.Z.) has led to the development of guidelines for constructed wetlands for farm dairy wastewaters (Tanner & Kloosterman, 1997). It is hoped that specific guidelines can be developed for the Australian dairy industry.

ACKNOWLEDGEMENTS

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DESIGN OF CONSTRUCTED WETLANDS FOR STORMWATER POLLUTION CONTROL

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ABSTRACT

Constructed wetlands are becoming increasingly popular as stormwater quality Best Management Practices (BMPs). When designed and maintained properly, these aesthetically appealing BMPs can be extremely effective for both flood attenuation and pollutant removal.

Constructed wetlands retain, detain and influence stormwater runoff enabling the pollutants to filter and settle or be taken up by natural processes before discharging into receiving water bodies. Pollutant removal efficiency and the relative importance of various pollutant removal pathways will vary depending on the nature of stormwater characteristics and the hydrodynamics within the wetland system.

The performance of constructed wetlands depends greatly on their design. Factors that must be considered in the design process for successful pollutant removal include the unsteady, intermittent nature of stormwater characteristics, hydraulic retention time, hydraulic efficiency, basin shape, groundwater, vegetation and inlet/outlet structures.

Wetland vegetation performs a number of important roles in constructed wetlands including physical, biological and chemical functions. These may range from energy dissipation and flow redirection to nutrient uptake. Vegetation can have a significant impact on the hydraulic retention time within a wetland by altering its hydrodynamics and thus enhance the primary pollutant removal pathway of sedimentation.

For optimum pollutant removal effectiveness, the stormwater characteristics, wetland hydrology and hydrodynamic characteristics must be considered in the design process under a range of operating conditions.

INTRODUCTION

The success of constructed wetlands in the wastewater treatment industry and their potential to be incorporated into the environment as a landscape feature has resulted in an increase in the use of these Best Management Practices (BMPs) for stormwater treatment. The concept of stormwater treatment using constructed wetlands utilises similar design features and processes used to treat wastewater. However, these stormwater treatment facilities must be able to deal with the intermittent and unsteady hydraulic and pollutant loads (Wong, 1997a). The design of constructed wetlands for stormwater treatment is an evolving field and design criteria will be refined as the results of performance monitoring become available.

POLLUTANT REMOVAL

MAJOR POLLUTANTS

Stormwater pollutants are highly varied in their composition as they are sourced from catchments with very different characteristics. Table 1 presents information collated by Kadlec and Knight (1996) on the composition of stormwater and summarises typical pollutant concentrations and ranges.

Table 1 - Stormwater Composition (adapted from Kadlec & Knight, 1996)

CONSTITUENT	CONCENTRATION mg/l
BOD_5	20 (7-56)
COD	75 (20-275)
TSS	150 (20-2890)
VSS	88 (53-122)
NH_3-N	0.59
TKN	1.4 (0.6-4.2)
TN	2 (0.7-20)
Ortho-P	0.12
TP	0.36 (0.02-4.3)

Most constituents in stormwater are time dependent and cyclic concentration patterns are typical. This is due to periods of dry fall and deposition, followed by the first flush and the exponential decrease in concentrations as pollutants rinse from the catchment (Kadlec & Knight, 1996).

POLLUTANT REMOVAL PROCESSES

The treatment train approach becoming popular with regulatory authorities encourages the use of pollution control measures throughout the catchment so that end of line measures are not solely relied upon for pollution control. This approach encourages the implementation of a number of management measures placed in series or concurrently throughout the catchment. The sequence of the various processes is an important design consideration as upstream pollutant removal will influence the performance of treatment measures downstream.

The treatment processes that occur within wetlands are a complex combination of interacting physical, chemical and biological mechanisms (Wong, 1997a,b). Physical processes are usually enhanced in the initial stages of wetland systems as it is often required to pre-treat the stormwater in order to optimise the performance of the chemical and biological pollutant removal pathways.

The optimum wetland design will depend on the composition of the stormwater to be treated. If the stormwater has a large proportion of particulate matter, wetland design should focus on physical processes such as sedimentation. However, if there is typically a high proportion of dissolved pollutants in the stormwater, the wetland should be designed to enhance the biological and chemical treatment processes.

The hydrological conditions and the stormwater composition determine the relative importance of the various treatment processes. During baseflow conditions, the most important mechanisms of pollutant control are biological and chemical transformations. The longer detention times associated with the lower baseflow conditions allow more time for biological and chemical processes to occur. Event flow conditions occur when storm events increase the flow volumes. This reduces the hydraulic detention time so physical pollutant removal mechanisms such as sedimentation will dominate.

DESIGN CONSIDERATIONS

Stormwater quantity was the only focus of drainage design in the past and it was only recently that measures for stormwater quality were considered important. Thus a movement from the traditional conveyance approach to detention and retardation systems retrofitted for use as both stormwater quantity and quality management has been seen. Even though there has been an increasing trend in the use of existing and constructed wetlands as landscape features in urban developments, their potential for stormwater quality treatment has not been optimised. This is reflected by their poor design in relation to the lack of an integrated approach in their design, operation and management (Lloyd et al., 1997).

Wong (1997a) states that a lack of appreciation for the treatment train approach and poor wetland hydrodynamics has led to a number of problems encountered with constructed wetlands. These problems have included:

- litter, oil and scum accumulation in some sections of the wetland;
- infestation of weeds or dominance of certain types of species of vegetation;
- mosquito problems;
- algal blooms;
- sections of the wetland subject to scouring and ultimate mobilisation of trapped pollutants.

Due to the complex interactions that occur within wetlands, their design requires a multi-disciplinary approach which allows for appropriate integration of many forms of input (White, 1996). Biological, chemical, ecological, hydrological and hydraulic components need to be integrated for optimum stormwater treatment outcomes. Wong (1997b) identifies inputs that should be considered in the design process including the catchment and wetland hydrology as well as the hydrodynamic behavior within the system. The effect that these factors have on the morphological and botanical features of the wetland need to be examined so that relationships between them can be determined.

STORMWATER CHARACTERISTICS

The wide range of conditions that stormwater treatment facilities operate under has made it difficult to predict their performance and to derive specific design guidelines for them. The effectiveness of constructed wetlands at pollutant removal is influenced by a number of non-linear factors including the relative difference between the influent and background pollutant concentrations and the flow hydrodynamics (Kadlec and Knight, 1996).

Sizing of wetland systems should take into account pollutant concentrations. There is likely to be a significant amount of intra-event variability in concentrations in stormwater but a clear understanding of the stormwater to be treated is essential for effective pollutant removal (Wong, 1997b). In this way, the treatment system can be designed to enhance the most significant pollutant removal pathways.

HYDROLOGIC CHARACTERISTICS

The most important aspect to consider when designing a constructed wetland is the system hydrology. The seasonality of stormwater flows must be incorporated into the design so that the wetland functions appropriately under the variable hydraulic conditions. In addition to monthly flow estimates, stormwater treatment systems require a definition of the frequencies of events and their magnitude and timing (Kadlec & Knight, 1996).

Fluctuations in the hydraulic and pollutant loadings complicate the water treatment processes. Stochastic elements pointed out by Kadlec and Knight (1996) which lead the fluctuation of these factors include:

- influence of precipitation, evapotranspiration and infiltration on the background concentration and the hydraulic loading of the wetland;
- temperature and seasonality on the biological mechanisms in the wetland.

Stormwater characteristics discussed by Wong (1997b) which further increase the level of stochasticity in hydraulic and pollutant loadings include the following:

- unsteady intermittent nature of runoff resulting from variability of rainfall depth, storm duration and storm temporal pattern;
- unsteady intermittent nature of pollutant loading resulting from the inflow behavior outlined above;
- variability in the rate of pollutant accumulation during the period preceding a storm event and variable time-distribution of pollutant concentration during a storm event; and
- different time-distribution characteristics of pollutant concentration depending on the water quality parameter in question during a storm event.

One of the major operations of the treatment sequence in a wetland is the attenuation of the unsteady inflow to a condition that approaches steady flow. The effectiveness of stormwater treatment by detention is most significantly affected by the antecedent water level in the system as this influences the attenuation of the inflow (Wong, 1997a). The available detention storage volume, the emptying rate of the detention system and period between storm events will all influence the antecedent water level immediately prior to the occurrence of inflows.

Wong et al. (1996) derived interaction charts for constructed wetlands in Southern Western Australia highlighting the inter-relationship between the following parameters:

- detention period;
- the volume of wetland storage available for detention; and
- the Hydrologic Effectiveness.

Wong and Somes (1995) point out that for a given size wetland, these three parameters are interrelated as the Hydraulic Effectiveness, defined as the overall percentage of runoff which can be expected to be detained at or longer than the desired detention period under intermittent loading conditions, varies inversely with the detention period.

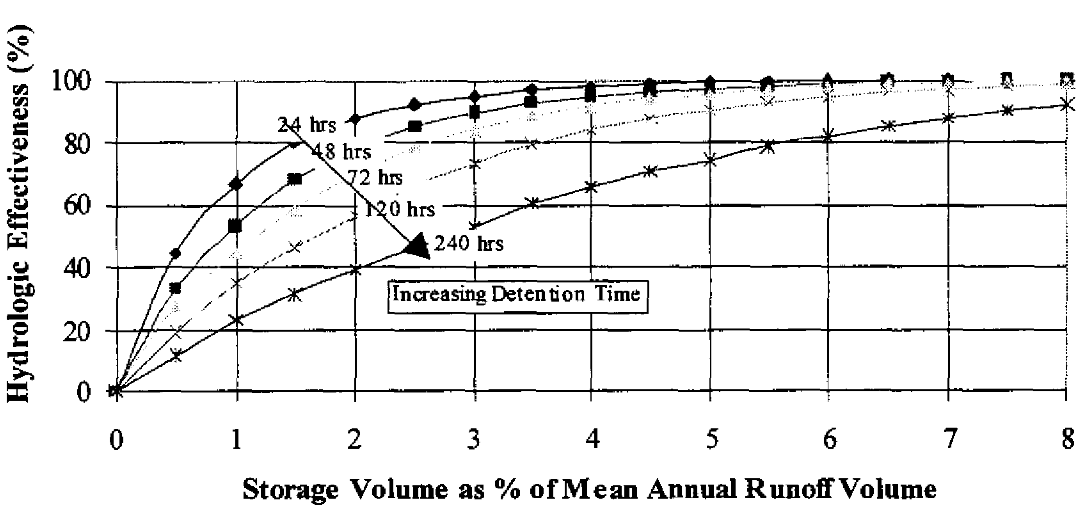

Figure 1 - Hydrological Effectiveness of Constructed Wetlands in Perth (Wong et al., 1996)

Figure 1 shows the results of wetland behavior simulations performed by Wong et al. (1996) for constructed wetlands in Perth. A range of storage volumes and prescribed minimum detention times were analysed and plots of the percentage runoff treated (i.e. Hydrologic Effectiveness) versus storage volume obtained for each prescribed minimum detention period developed. These plots are incorporated into the design guidelines in the Draft Stormwater Quality Management Manual prepared by Evangelisti & Associates in association with Tony Wong & Associates and Alan Tingay & Associates for the Water & Rivers Commission. They will allow the selection of appropriate storage volumes and outlet characteristics to be based on the long term average effectiveness of the wetland in retaining stormwater over a desired minimum period rather than a prescribed level of performance for an individual probabilistic event.

Wong and Somes (1995) also prepared similar Hydrologic Effectiveness curves to Figure 1 for Melbourne. Curves for individual regions will be different owing to differences in the characteristics in their respective rainfall intensity-frequency-duration relationships, seasonal rainfall distribution, rainfall duration and inter-event dry periods (Wong, 1997a).

The first flush of pollutants conveyed by storm events during the early periods of autumn in the months of March and April in south Western Australia requires additional consideration and special treatment. Sufficient storage should be available for the first flush which generally contains higher pollutant concentrations due to the build up in the catchment over the extended dry period prior to the first flush.

In order to provide sufficient storage for the first flush, the lowest level of a riser outlet should ideally be placed at the Average Annual Maximum Groundwater Level (AAMGL). The system should be designed so that the drawdown in groundwater levels over summer can provide sufficient storage for the first flush. The water levels in the constructed wetland are expected to fluctuate in a similar manner as the groundwater level fluctuation between its design permanent pool and the Average Annual Minimum Groundwater Level (Wong et al., 1996).

Local groundwater conditions must be investigated to ascertain the available retention storage provided by the expected difference in the pool level at the commencement of Autumn and the design permanent pool of the wetland. The composition of the groundwater should also be investigated as it may become a pollutant source if not designed appropriately. In Perth conditions, the wetland design should be checked against the probabilistic runoff volumes from the catchment for the months of March and April. This can then be compared with the volume provided by the wetland between the design permanent pool level and the AAMGL to determine if additional storage volume must be designed for. The Stormwater Quality Management Manual be published by the Water & Rivers Commission (Perth) describes this design process in more detail.

HYDRODYNAMICS

The design of a constructed wetland should aim for a uniform distribution of flow so that stormwater treatment is maximised and short circuiting reduced. The distribution of flow can be influenced by a number of key factors. These include the following and are discussed further below:

- Shape;
- Compartmentalisation;
- Vegetation; and
- Hydrologic Regime.

The length to width ratio of the wetland basin has a significant effect on the flow distribution and hydraulic short-circuiting. Kadlec & Knight (1996) recommend a minimum length-to-width ratio of 2:1 which balances the enhanced effluent distribution obtained by higher aspect ratios with the increased cost of earthworks. Additional methods for flow redistribution, such as deep zones, should also be incorporated into the design to reduce the need for higher length-to-width ratios (Kadlec & Knight, 1996). As discussed by Kadlec & Knight (1996), inlet and outlet locations should be placed so that the flow path is maximised and thus treatment optimised. Multiple cells in series should also be considered as these can further distribute the flow and reduce short circuiting.

Vegetation

Somes et al. (1996) reviewed the functions of wetland vegetation during baseflow and eventflow conditions. The different detention times that occur during baseflow (ie. periods between runoff events) and eventflow situations results in the changing role that vegetation plays. Shorter detention times during eventflow conditions reduce the significance of biological and chemical processes (Wong, 1997a). Table 2 presents the various treatment functions of vegetation in constructed wetlands.

The importance of the physical processes that enhance sedimentation are very important during eventflow conditions as the majority of pollutants are transported during storm events. Trapped sediments and their associated pollutants will undergo further biological and chemical treatment processes during the intervening baseflow periods (Wong, 1997a).

Table 2 - Treatment Functions of Vegetation in Constructed Wetlands for Stormwater Control (Somes et al., 1996)

Baseflow	Eventflow
Act as substrata for epiphytes (Ephiphytes convert soluble nutrients into particulate biomass	Promote even distribution of flows
that can settle out and enter the sediments-short term process)	Promote sedimentation of larger particles
Consolidate nutrients trapped in the sediments into macrophyte biomass (medium-term process)	Provide surface area for adhesion of smaller particles
Return particulate biomass as macrophyte litter for storage in the sediments (long term process	Protect sediments from erosion
resulting in the development of organic sediment and peats)	Increase systems hydraulic roughness

Vegetation is involved in a number of treatment processes including:

- energy dissipation and enhanced sedimentation;
- flow redirection;
- fine particle filtration;
- nutrient uptake;
- nutrient storage in living and dead plant biomass;
- soil stabilisation; and
- control of the redox potential in the sediment by supply of oxygen through the root system.

Vegetation plays an important role in the dissipation of kinetic energy by the reduction of linear velocities and subsequent sedimentation of suspended solids. Dense stands of vegetation will help to reduce flow velocities and increase the degree of treatment the stormwater is subjected to. Wong (1997a) recommends that a bypass system be incorporated in the inlet zone so that large events may be diverted away from the wetland system. This will reduce the potential for scour and resuspension of settled material in the wetland if linear velocities result in excessive values of shear stress.

In order to maximise wetland treatment performance, zones of wetland vegetation associated with desired functions need to be created. The plants need to be ecologically adapted to the hydrologic regime and have suitable morphologies so that treatment is optimised. Ideally, these zones should be arranged in series across the flow path but the design of individual wetlands will vary depending on the local topography and site conditions (Wong, 1997a).

Hydrologic Regime

Wong (1997a) defines the hydrologic regime as the probabilistic distribution of the depth of inundation within the wetland. The hydrologic regime affects the distribution of aquatic plants within the wetland and is defined by the hydraulic characteristics of the inlet and outlet structures. The system hydrology will determine the inundation frequency, duration and seasonality (Wong, 1997a). Somes et al. (1996) showed that a satisfactory hydrologic regime requires proper outlet design so that the desired vegetation layout is maintained. Figures 2 and 3 illustrate the differing hydrologic regimes simulated by Wong (1997a) for a typical wetland in Melbourne with different outlet structures. It is clear that outlet structures will have a significant impact on the complexity of the vegetation species and thus on the various pollutant removal processes that will occur within the wetland.

The proper control of the hydraulic regime involves a combination of providing adequate detention time for such processes as sedimentation and filtration to be effective and draining the wetland at a rate which would result in sufficient available storage to treat at least the first flush of the next runoff event (Wong et al., 1996).

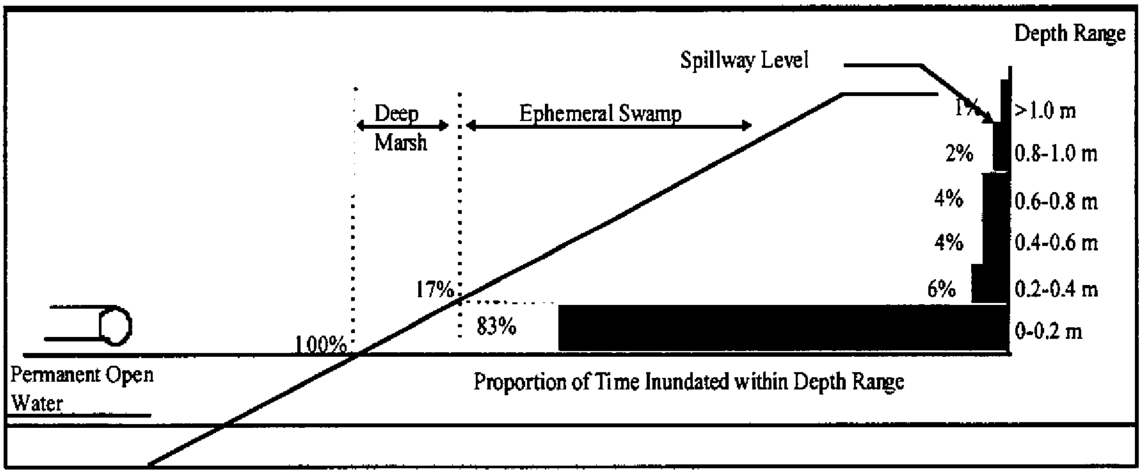

Figure 2 Hydrologic Regime of Constructed Wetland with Riser Outlet Control (Wong et al., 1996)

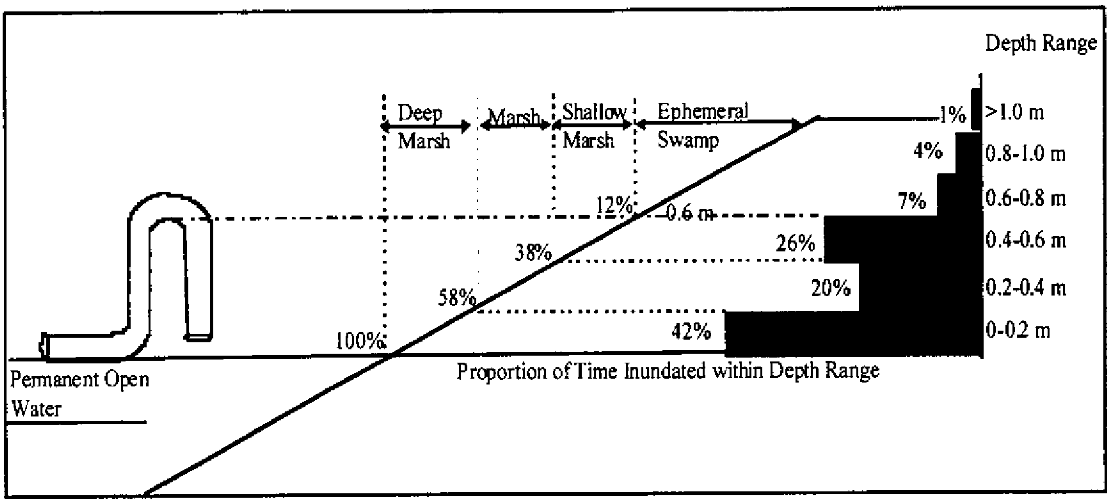

Figure 3 Hydrologic Regime of Constructed Wetland with Siphon/Glory Hole (Wong et al., 1996)

MONITORING

The effectiveness of constructed wetlands will vary from one storm event to another due to the unsteady and intermittent nature of the runoff hydrograph and associated pollutographs (Wong, 1997a,b). For useful data collection and analysis, a thorough understanding of the hydraulic residence times of both the water and the pollutant phases needs to be obtained and a sampling program that enables reliable estimates of flow weighted mean pollutant concentrations should be prepared (Lloyd et al., 1997). The difficulties in tracking individual "parcels" of water render it impractical to define the removal efficiencies of a constructed wetland at discrete time intervals (Lloyd et al., 1997).

In order to gain a better understanding of why different systems behave differently in their pollutant removal ability, we need to document the pollutant characteristics and hydrodynamic behavior of the wetlands for individual events. In order for better interpretation and comparisons to be made between data sets, Lloyd et al. (1997) recommend that the following information should be recorded:

- background pollutant concentration levels;
- input concentrations for each event and corresponding pollutant reduction figures;
- hydraulic loading; and
- other detailed information on the monitoring program.

CONCLUSION

The design philosophy of constructed wetlands is based on existing wastewater treatment technologies. However, the intermittent and unsteady hydraulic and pollutant loads associated with stormwater should be considered in the design process to ensure optimal treatment. The pollutant removal performance of constructed wetlands will be determined by a combination of the stormwater characteristics, system hydrology and hydrodynamics. Sustainable operation of these stormwater treatment facilities requires ongoing maintenance and monitoring to ensure that they operate at their full potential.

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THE DESIGN OF AN ONSITE SEWAGE TREATMENT SYSTEM UTILISING A FINE SAND FILTER AND A CONSTRUCTED GRAVEL REED BED TO TREAT SANITARY WASTEWATER

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ABSTRACT

Population growth and the economic changes in Botswana's rural areas create a demand for water and that in turn increases the problem of wastewater disposal. The rural areas in Botswana are characterised by low-density settlements and cannot therefore, benefit from the economies of scale that are possible with centralised wastewater management facilities provided in urban areas. The development of new institutions in most rural areas will in the foreseeable future continue to depend on onsite wastewater treatment and disposal systems. Inappropriate sanitation systems cause public health problems. It is common in the developing world for untreated wastewater to pass into water aquifers. This paper describes the design of an onsite sewage treatment system for the offices of Thusano Lefatsheng in Botswana. The system is made of a fine sand filter and a gravel reed bed wetland. Thusano Lefatsheng offices are located near a shallow aquifer and the paper compares the design solution with alternative systems. The construction of the system is complete and, once in function will be monitored, reviewed and developed further.

INTRODUCTION

Water pollution due to the use of inappropriate methods of wastewater disposal and inadequate treatment systems is beginning to be noticed in Botswana because of the increase in water use by industries and communities. Studies (WLPU, 1990) document surface and groundwater pollution in the form of nitrates and biological contamination. Water pollution is state primarily in the Water Act, which prohibits discharges that result in the pollution of public water and establishes sanctions. The Water Apportionment Board administers a system of discharge permits. Monitoring is done by the Department of Water Affairs, which reports violation to the Board. The Department of Water Affairs has issued several pollution control guidelines for specific aquifers and certain purposes. These sets of guidelines are not legally enforceable but aim to guide activities by providing information and persuasion.

Local Authorities under the Ministry of Local Government Lands and Housing are responsible for the provision of sanitation services infrastructure and operation and maintenance of the systems. Provision of excreta disposal facilities in the rural areas was the responsibility of the individual households until 1980 when USAID funded a pit latrine pilot construction programme in two districts. In 1984 the Botswana Government with the assistance of UNICEF then funded a subsidised pit latrine programme as an extension of the USAID project and the latrine construction has since developed into a national programme covering all districts. However, according to the 1991 Population Census Data the population covered with sanitation facilities nationally is 55% and the coverage is poorest in the rural areas where some Districts are as low as 7% (Central Statistics Office, 1991). In rural areas 28% and 5% of the population is served with sanitation facilities that use dry and water borne systems respectively.

The most common method of wastewater treatment and disposal in areas without centralised wastewater management has been the septic tank and disposal field system where feasible. The potential pollution due to the proliferation of the septic tank and disposal field system is beginning to cause concern. Other wastewater treatment facilities such as oxidation ponds also used in Botswana mainly for some government institutions do not achieve the treatment efficiency expected (WLPU, 1990). Pit latrines which are mainly used in the rural areas and by the urban low-income population are reported to be causing water pollution in areas of poorly drained soil or where they are underlain by fractured bedrock which allow the rapid transport of the contaminants to the groundwater sources.

THUSANO LEFATSHENG PROJECT WASTE WATER TREATMENT SYSTEM

Project Description

The scope of works for the Thusano Lefatsheng Project comprised the following:

- Supply and installation of approximately 350 metres of potable water supply to the administration office, kitchen, greenhouse, research facilities and 15 staff houses including connection to existing supply.
- Construction of approximately 250 metres of engineering earth road including lined V-drains, drifts and access culverts.
- Supply and installation of approximately 350 metres of reticulated sewer line to administration office, kitchen, research facilities and 15 staff houses including manholes, septic tanks and secondary sewage treatment system works.

The Botswana Technology Centre was contracted by Thusano Lefatsheng to provide the consultancy services for the architectural and civil engineering design and the construction supervision for the project. The design commenced late 1996 and the contractors started the construction early 1997. The civil engineering contractor completed the civil engineering aspect of the project at the end of July 1997, and the building contractor completed building the works early October 1997.

Sewage Flow Rates

The Thusano Lefatsheng site is located in Mankgodi village, approximately 25 km from the capital city Gaborone, along the main road to Jwaneng town. The administration offices, kitchen, greenhouse and research facility are located west of the main road. The staff houses are located east of the main road.

The average daily flow rates for the sewage were estimated as follows:

a) Administration offices, kitchen and research facility
40 workers x 41 litres/person/day = 1 640 l/d
= 1.64 m^3/d

b) Staff houses
15 x 4 person/house x 82 l/p/d = 4 920 l/d
= 4.92 m^3/d

An infiltration rate of 30% was considered because the groundwater table seems to be high and some areas of the site become very saturated during the rainy seasons.

Treatment System Design, Construction and Expected Performance

Staff Houses Site
At the staff houses site the difficult conditions on site were the possible high groundwater table and the seasonal soil saturation experienced during the rainy season. These site conditions ruled out the use of a septic tank and field disposal system.

At the staff housing site a centralised septic tank and constructed reed bed treatment system was adopted. The system was located on an area which suffers less season soil saturation. To avoid cutting down trees a long (64m) narrow (3m) reed bed 300mm deep was provided. The reed bed is located within the site approximately 20 metres from the nearest houses. When compared to other treatment systems such as oxidation pond and biofilters the reed bed is likely to cause less smell because of the cover by the reeds. A reed bed treatment system was therefore, preferred. To ensure that the effluent from the septic tank will easily flow throughout the length of the reed bed clean gravely sand media was used. The bottom and sides of the reed bed are lined with 250μm plastic sheet. The slope of the reed bed towards the outlet is 0.5%. The detention period for the effluent is not less than 6 days. From the outlet the effluent goes to a soak pit.

With a shallow reed bed aerobic life will flourish resulting in the oxidation of filtered solid, material in solution and the nitrification of organic matter. The long detention period will afford the system time to undergo the denitrification of the nitrates using carbon from the decaying leaves of the reeds. The treatment process will remove BOD to less than 20 mg/l. Nitrates and biological contaminants removal will also be significant. However, it is difficult to estimate the extent of nitrates and biological contaminants removal. The system will be monitored when it starts functioning and empirical data will be developed on the performance of the system. The cost of the reed bed construction was US $4300 at 1997 prices in Botswana.

Administration Office, Greenhouse, Kitchen and Research Facility Site
The difficult conditions on this site were the possibility of high groundwater table and the fact that the borehole which supplies the water used at both sites is on this site. Location of the buildings was dictated by the land which was allocated for the development and the need to locate the sewage treatment system as far as possible from the borehole. The treatment system was located approximately 400m from the borehole.

Fig.1 below shows the wastewater treatment system which was adopted for this site.

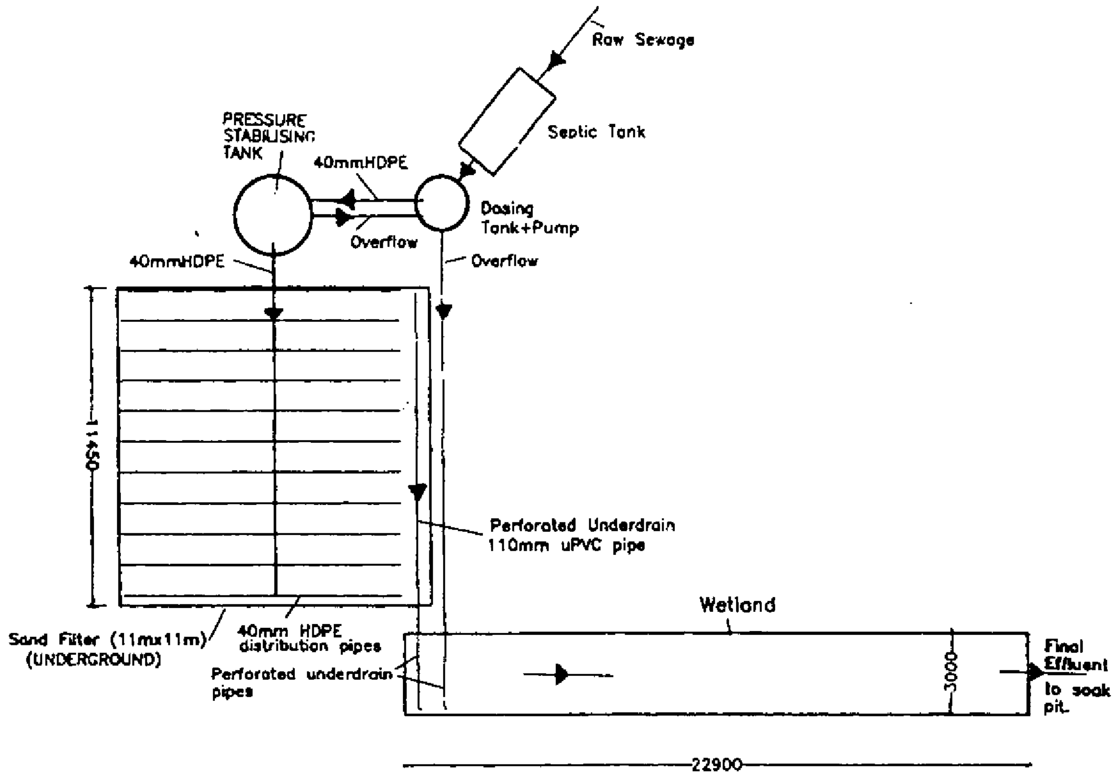

FIG. 1 TREATMENT SYSTEM LAYOUT

The treatment system comprises the following components:

Septic Tank: To provide the primary treatment and storage for settleable solids for a period of 5 years before the sludge can be pumped and taken by Local Authorities to approved landfill sites.

Dosing Tank: This is the tank from where the effluent from the septic tank is pumped to the pressure stabilising tank. From the dosing tank there is also provision for effluent to overflow directly to the reed bed during the maintenance of the pump and the sand filter. The overflow to the reed bed is also meant to inject carbon rich effluent for the denitrification of nitrates in the effluent coming out of the sand filter. The dosing tank is built from ferrocement material. The tank diameter is 1900mm and the total height is approximately 1.5 metres including free board. If the pump is not working the tank will store about one and half days effluent before it can overflow. The dosing tank cost was US $900 which is about eight percent of the total cost of the system excluding the septic tank.

Pump System: A submersible solar pump was chosen for the project in order to minimise maintenance and operation costs. The use of a solar pump system is environmentally friendly and ideal for Botswana because of the long sunshine hours. The pump system including installation cost was approximately US $4300 which is approximately 36% of the total cost of the system excluding the septic tank.

Pressure Stabilising Tank: Effluent pumped from the dosing tank to the pressure stabilising tank flow under natural head to the sand filter. The pressure stabilising tank is provided to ensure that the head supplying effluent to the sand filter remains within a range of 1.5 to 3m for the optimum performance of the sand filter. The distribution system piping was sized so that the discharge from each orifice in the distribution system is nearly the same as possible. The equal flow from orifices was achieved by ensuring that the head loss in the distribution pipes is low as compared to the head loss through the orifices. The pressure stabilising tank is built from ferrocement material. It has provision for the effluent to flow back to the dosing tank in the event the sand filter is overloaded and cannot take any more. The pressure stabilising tank cost was US $700 which is about six percent of the total cost of the system excluding the septic tank.

Sand Filter: The sand filter is approximately 500mm below ground level. Because the septic tank, dosing tank and pressure stabilising tank are all covered and the sand filter is underground, there will be no problem of smell. The bottom and sides of the sand filter are lined with two layers of 250μm plastic sheet. One layer of 250μm plastic was placed on top of the sand filter before backfill to stop the backfill soil etc from contaminating the filter material.

Inside dimensions for the sand filter are 11m x 11m and 800mm deep. A 110mm diameter perforated underdrain pipe is provided on one edge of the filter to drain the treated effluent out of the sand filter. To reduce the sand which migrates to the underdrain pipe a 1200mm wide strip of 13mm crushed stone is placed on one side of the underdrain pipe throughout the height of the filter. The underdrain pipe is connected to a 110mm plastic pipe protruding 500mm above the ground. The 110mm pipe is closed with a cap which can be removed when the need arises to wash out the underdrain pipe. The rest of the sand filter is filled with fine sand to a depth of 600mm. A hydraulic loading of 0.5 cubic feet per square feet per day was used for the design of the sand filter. The uniformity coefficient and effective size of the sand used in the sand filter are 2.5 and 0.25 respectively. On top of the sand is placed a 100mm layer of 13mm crushed stone then the effluent distributing pipes are positioned on top of the stones. The stones help to further distribute the effluent on the sand filter. Another layer of 100mm crushed stone is placed on top of the distribution pipes before the plastic cover is placed. The plastic cover is then placed and the whole system closed up underground.

The effluent distribution pipes are connected to the pressure stabilising tank where the system is fed from. The distribution pipe consists of a 40mm pipe running in the middle with 40mm lateral pipes at 1200mm interval to take the effluent to the sides. Lateral pipes have 3mm holes facing up at 1200mm intervals to distribute the effluent evenly over the system. To avoid any effluent flowing straight into the underdrain pipe before treatment the lateral pipes stop 900mm before the edge of the crushed stone covering the underdrain pipe. The sand filter will achieve a high degree of treatment for BOD, suspended solids and coliform organisms. The sand filter will also nitrify the ammonia in the septic tank effluent. The sand filter cost approximately US $5000 which is about 42% of the total cost of the system excluding the septic tank.

Reed Bed: The effluent from the sand filter is discharged to a constructed reed bed. The bottom and sides of the reed bed are lined with 250μm plastic sheet. Clean medium size grain was used as the basin media for the reed bed. The slope of the reed bed towards the outlet is

0.5%. The detention period for the effluent is not less than 6 days. Also to the reed bed is an overflow from the dosing tank which can be used to take septic tank effluent which is still rich in carbon to the reed bed. The nitrates in the effluent from the sand filter are to be removed by denitrification process using a bit of septic tank effluent from the dosing tank as the carbon source. The decaying of the leaves of the reeds will also provide some of the carbon needed for the denitrification process. The reed bed will remove further the suspended solids and coliform organisms. From the outlet the final effluent goes to a soak pit. The reed bed cost approximately US $1000 which is about 8% of the total cost of the system excluding the septic tank.

CONCLUSIONS AND RECOMMENDATIONS

a. The use of a pressure dosed sand filter combined with a reed bed treatment system will achieve a high degree of treatment of the septic tank sewage effluent.

b. The cost of the pressure dosed sand filter combined with a reed bed is approximately US $11900 at 1997 prices in Botswana. The sand filter system and pump system contribute 42% and 36% of the total cost of the whole treatment system respectively. A reduction in the cost of the pump system or the sand filter will reduce the cost of the treatment system significantly.

c. The use of a narrow (3m) and long (64m) reed bed at the staff houses site will provide empirical information on the performance of such a design. The effect of a long detention period (6 to 10 days) will also be assessed.

d. To limit the failure of onsite wastewater treatment systems local codes or design guidelines based on local experience of potential onsite wastewater treatment systems should be developed because the local authorities are neither empowered nor funded to provide individual designs for onsite treatment systems in Botswana.

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Case Studies

ARTIFICIAL RECHARGE OF URBAN WASTEWATER, A VALUABLE WATER RESOURCE MANAGEMENT STRATEGY IN THE MORE ARID ENVIRONMENTS

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ABSTRACT

In many developing countries, water is a scarce and valuable resource. The demand for water is growing and scarcity of water is placing increasing limitations upon the options available for urban, industrial and rural development. There is therefore, a need to assess and manage the available water resources in a realistic and more efficient manner. Water reuse has long been recognised as one means of conserving water. The public, however, have a tendency to object to direct reuse of wastewater, although the artificial recharge of aquifers with reclaimed water is considered by many as more acceptable. The recharged water is seen to lose its identity in the groundwater mass and the direct connection between wastewater treatment plant and water supply is not as apparent.

Artificial recharge is being practised in a great many countries using either infiltration basins or injection wells. Atlantis in South Africa is, however, rather unique in that artificially recharged wastewater is used as a potable water supply. The town is totally dependant on groundwater resources from a local shallow, sandy aquifer for its water supply. A well designed stormwater drainage system and scientifically based water quality management strategy allows for the recharge of all the urban stormwater runoff and treated domestic sewage effluent. Extensive research and sound planning has meant that no water is wasted, as even that water which is unacceptable for recharge in the main aquifer is recharged in strategic localities to counter any saline intrusion. The Atlantis Water Resource Management Scheme (AWRMS) serves as a good example of what can be achieved when hydrogeologists, engineers and urban managers integrate their skills.

INTRODUCTION

Atlantis, an industrial town on the somewhat bleak west coast of South Africa serves as an example of how, even with limited water resources, urban development can be sustained. Situated some 50 km north of Cape Town, the town was to be supplied with potable water from the Berg River, approximately 100 km to the northwest (Figure 1). As this supply was not immediately available, water was taken from a local spring. Subsequent hydrogeological investigations showed that this was a surface outlet of a primary coastal aquifer, which if correctly exploited, could prove to be of great value in the development of the town.

Today, some 20 years after development commenced, the town has a population of over 67 000 and an industrial sector of 140 factories requiring a reliable supply of potable water in excess of 5.5×10^6 m^3 per annum. This demand is met by the Atlantis Water Resource Management Scheme (Figure 2) which utilizes both the local unconfined coastal aquifer and all wastewater originating from the town. Groundwater is extracted from the aquifer at two well fields, treated in an ion-exchange water softening plant, distributed, utilized, collected, treated and artificially recharged together with urban stormwater runoff back into the aquifer. Maximum use is thus made of the limited water resources. The current state of water supply has been achieved by careful exploration and development of the aquifer, coupled with sound management of both the resource and urban development.

THE ATLANTIS WATER RESOURCE MANAGEMENT SCHEME

The scheme consists of a number of different components (illustrated in Figure 3) and may be summarized as follows:

The aquifer

The area is characterised by extensive deposits of Cenozoic sediments that are underlain by shales and greywackes of Late Proterozoic age (Malmesbury Group). The upper surface area of the aquifer is covered by either mobile sand dunes or consolidated vegetated sands. The unconsolidated Cenozoic sediments (largely silica sands) constitute an unconfined aquifer and consists of two stratigraphic formations. The lower Varswater Formation is of shallow marine origin and the younger Bredasdorp Formation of aeolian origin. The saturated thickness of the aquifer varies considerably but it seldom attains more than 35 m in the areas where abstraction takes place. The bedrock topography is responsible for creating five compartments/sub-units within the aquifer and is to a large extent related to the ancient drainage system and to raised beaches. In two small areas the base of the aquifer extends below sea level. Inland the aquifer pinches out against outcrops of the Malmesbury Group or Cape Granite Suite. In general, the aquifer is inhomogeneous, anisotropic and phreatic. Groundwater bodies of different salinities can be traced laterally and occasionally vertically. Natural recharge occurs over the whole of the aquifer, but appears to be more pronounced in a large area of shifting dune sand where the natural precipitation filters directly into the aquifer (Tredoux & Tworeck, 1984).

The production wellfields

Hydrogeological investigations identified two promising areas for the abstraction of good quality groundwater. Extensive drilling took place and today the aquifer contains exploration, production and monitoring bores. Groundwater is abstracted from two major wellfields, namely Silwerstroom and Witzand (Figure 3). Both well fields are still being extended as the demand for water increases. The groundwater quality in the southern portion of Witzand wellfield has a relatively high total hardness, caused by the presence of calcium and magnesium bicarbonates. Even when blended with better quality water from other parts of the aquifer, it has to be softened in an ion exchange water softening plant before use in the industrial area. The softening plant, a weak base ion exchange plant using sulphuric acid as a regenerant, generates an acidic calcium sulphate and magnesium sulphate waste which is diluted with treated industrial effluent before being disposed of in a series of infiltration basins on the coast. Groundwater quality and groundwater levels are closely monitored to ensure that the correct abstraction policy is followed. Fully automated flow gauging and water-level recording devices are connected to a central control station via a telemetry network.

Sewerage reticulation system

Atlantis has twin sewerage reticulation systems which allow for separation of sewage from the residential and industrial areas and corresponding parallel waste water treatment works. The treated domestic effluent, although containing relatively high concentrations of nutrients (Table 1) does not generally contain the large number of chemical and synthetic pollutants found in industrial effluent and is considered re-usable, either for artificial recharge or irrigation purposes. The domestic sewage is treated in an activated sludge works. Preliminary treatment involves screening and grit removal after which the influent is pumped using an Archimedes screw pump for further treatment in anoxic zones and the main aerated reactor. A fine bubble diffuser system is used and a secondary sedimentation tank separates the sludge to be returned to the anoxic zones. A separate screw pump returns the mixed liquor to the anoxic zones for denitrification. After passing through three maturation basins, the final effluent is discharged into stormwater basin 6 and blended with the stormwater before being discharged into the poorer quality recharge basin. The treated industrial effluent, together with the water softening plant brine, is not considered re-usable for town supply and is disposed of by means of recharge in a series of coastal basins.

The stormwater system

Under natural conditions very little surface runoff is generated in the Atlantis area. This is borne out by the virtual absence of natural surface drainage systems. The fact that urban development would give rise to appreciable volumes of stormwater led to the design of a system of detention and retention basins for stormwater collection and disposal. The system consists of underground pipes and twelve detention basins. The detention basins are designed to reduce peak flows and those in the industrial zone have pollution reduction features such as reed-beds and sediment traps. The system has a series of diversion mechanisms allowing for the separation of poor quality stormwater away from the artificial recharge basins and into either the Donkergat Creek or the coastal recharge basins. The system is carefully monitored to ensure that only water of an acceptable quality enters the artificial recharge basins.

Initial studies indicated that the sandy nature of the catchment, small percentage area of effective impervious surface and high groundwater table resulted in the baseflow constituting more than 40 % of the total stormwater runoff and accounting for over 60 % of the pollution load. In many areas the stormwater drainage system intersects the groundwater table and groundwater drains into the system via the unsealed connections between pipe lengths. The stormwater runoff has very definite spatial quality variations which reflect both land use and groundwater quality. Different industrial and construction activities act as distinct point sources and pose the greatest threat to the stormwater runoff quality. The seasonal trend, reflected by a number of water quality variables, is linked directly to the stormflow/baseflow ratio within the stormwater runoff. Although a first flush effect occurs during storms, this is of limited significance due to the higher salinity of the baseflow. This lack of significance is a reflection of the large amount of rapid interflow compared to overland flow in the stormflow component.

During storm events the stormwater discharge may be as high as 72 000 m^3 day^{-1} while baseflow discharge during the dry months averages 2 160 m^3 day^{-1} (Wright, 1991). The industrial stormwater runoff has a relatively high background salinity and is susceptible to spills, leakages and illegal flushing. As such, the baseflow is not considered fit for artificial recharge upgradient of the wellfield. Stormwater pond 10, which serves the noxious trade area, is completely isolated from the main stormwater system and discharges directly into the coastal recharge basin outfall pipeline.

The artificial recharge basins

Two large basins situated in the dunes to the south of the town serve as final retention ponds and provide for the artificial recharge of the aquifer some 500 m up-gradient of the Witzand well field. Three sources of water are available from the urban catchment for recharge purposes, namely, stormwater runoff, groundwater and treated wastewater. The stormwater system collects both stormwater runoff and groundwater drainage, while the dual sewerage system divides the wastewater into industrial and domestic components. Table 1 summarises the water quality of the respective components. A number of different recharge options are thus available. Between 1985 and 1989 a number of different management options were implemented in order to define the best management strategy. This exercise was carried out at the more southerly recharge basin (Figure 3) and the volumetric effects are summarised in Figure 4. During 1985 and 1986 the basin received the maximum available water, whereas in 1988 it received only the best quality water and hence a greatly reduced volume of water. The 1989 recharge strategy was a better compromise and by 1990 the current strategy, as outlined in Figure 3, was in place. It was found that a certain volume of water has to be recharged in the poorer quality recharge basin in order to counter the flow of more saline groundwater from the south-east towards the southern portion of the Witzand wellfield.

Table 1: Quality of the water available for recharge and that of the aquifer

Determinant	Stormwater		Sewage effluent		Aquifer
	Residential	Industrial	Domestic	Industrial	
K	7.8	5.2	21.0	14.0	5.6
Cl	161	236	242	290	116
SO_4	56	83	98	382	41
NH_4 as N	0.1	0.1	0.2	0.1	<0.1
NO_x as N	4.3	1.5	1.4	12.6	0.3
PO_4 as P	<0.1	0.1	0.3	5.1	<0.1
DOC	10.3	36.5	12.4	13.4	8.4
EC (mS/m)	84	118	119	150	67

Units = mg/L, unless otherwise indicated

The different management strategies produced distinct recharge water quality signatures which could be detected in the groundwater quality. The basic recharge criterion established at the start of the project was that the acceptability of water for artificial recharge should be determined by the water quality of the receiving groundwater body/ aquifer. The effect which the recharge water had on the surrounding aquifer was carefully monitored in 97 observation bores and piezometers and analysed for up to 18 different water quality variables. A direct correlation could be made between the groundwater quality trends and that of the water in the recharge basin. Figure 5 shows the groundwater trends (in this case chloride) in two bores 80 m (WP20) and 200 m (WP15) downgradient of the recharge facility. Recharge cycles can be clearly identified and a delay time measured between the two. The fact that variables such as potassium could be traced over substantial distances suggested that the purification processes during infiltration were not as effective as expected. This was considered to be due to the low clay content of the sands and the fact that for much of the year the groundwater mound intersected portions of the basin floor. Where clay lenses were present the potassium was effectively removed.

It was also found that the infiltrating water did not immediately disperse vertically throughout the aquifer. Figure 6 shows how the influence from the recharged water was initially greatest in the upper portion of the aquifer, WP15 representing the upper part of the aquifer and WP14 the lower. With time, however, this hydrochemical stratification has diminished. It is only on the edge of the wellfield that mixing, caused by the production bores, results in a homogenous water quality.

It was concluded that several factors, including the nature of the sand and limited unsaturated zone, resulted in the purification process during infiltration not being as effective as was expected. Thus the most efficient method of improving the resultant recharged water quality was by controlling the quality of the water discharged into the basin. Although this reduced the volumes of water discharged into the basin and therefore the volume available for infiltration.

The coastal recharge basins

A series of infiltration basins near the coast are used for disposal of poorer quality wastewater from Atlantis. This includes waste originating from the regeneration of the softening plant ion exchange resins. It also includes treated industrial wastewater and the baseflow from the noxious trade area stormwater collection system. This system both provides an environmentally acceptable way of disposing of poorer quality water and also forms a barrier between the Witzand wellfield and any possible saline intrusion from the sea.

RESOURCE MANAGEMENT

The implementation of the AWRMS has provided Atlantis with the maximum benefit from its natural resource. Sound planning has meant that little water is wasted, as even that water which is unacceptable for recharge in the main aquifer is disposed of in a constructive way without polluting either the marine or terrestrial environments. Over the years, flexibility has been built into the system to enable management to include or exclude the various components as dictated by water quality needs and supply demands. An extensive monitoring programme acts as a safeguard in case of failure within any component of the scheme. The monitoring network includes: a meterological station, 5 flow gauging stations, 3 surface water level recording stations, 165 groundwater level recording sites, 15 stormwater monitoring sites, 4 waste water effluent monitoring sites, and 126 groundwater quality monitoring sites. Sampling is done on a weekly and monthly basis and involves both chemical and microbiological analysis. Flow gauging and water-level measurements are done on a continuous basis with the data being fed into a central data base. The database is a multi-user window-based database management system.

As the management programme evolved, it became necessary from time to time to test the impact of certain decisions on the aquifer, e.g. the influence of increased artificial recharge on the flow patterns of water or whether increased abstraction from certain bores would attract inferior quality water. A 2-D integrated finite difference model was developed in-house for this purpose. This model has subsequently been upgraded to a 2-D finite element model, AQUAMOD. A solute transport model using MT30, is currently being developed as an additional aquifer management tool.

The aquifer is further protected by means of effective land-use planning. Areas such as the Witzand wellfield and mobile dune fields have been set aside as nature reserves. The urban area with its many potential pollution sources is located outside of the good quality aquifer units that contain the recharge zones and wellfields. The scheme is constantly monitored and refined to meet any new demands thereby ensuring the long term viability of the AWRMS. The management team, which includes the consulting engineers, hydrogeologists and environmentalists, meets regularly to discuss the day to day management of the scheme. In addition independent consultants are employed to undertake an annual audit of the entire scheme.

CONCLUSION

It was clear from the outset of the project that innovative procedures would be required in order to establish and sustain urban development in this rather desolate terrain. It would necessitate a multi-disciplinary approach and close co-operation between different professional bodies. The outcome has been a highly cost effective water resource management scheme which can sustain continued urban growth well into the 21st century. The current cost of water is 20% of what it would have been had surface water been piped in from the Berg River. Without doubt one of the greatest successes of the project has been the manner in which several agencies have been able to co-operate in research and practical application of the results for the common good. Today, Atlantis serves as a prototype for further development in the more arid areas of Southern Africa.

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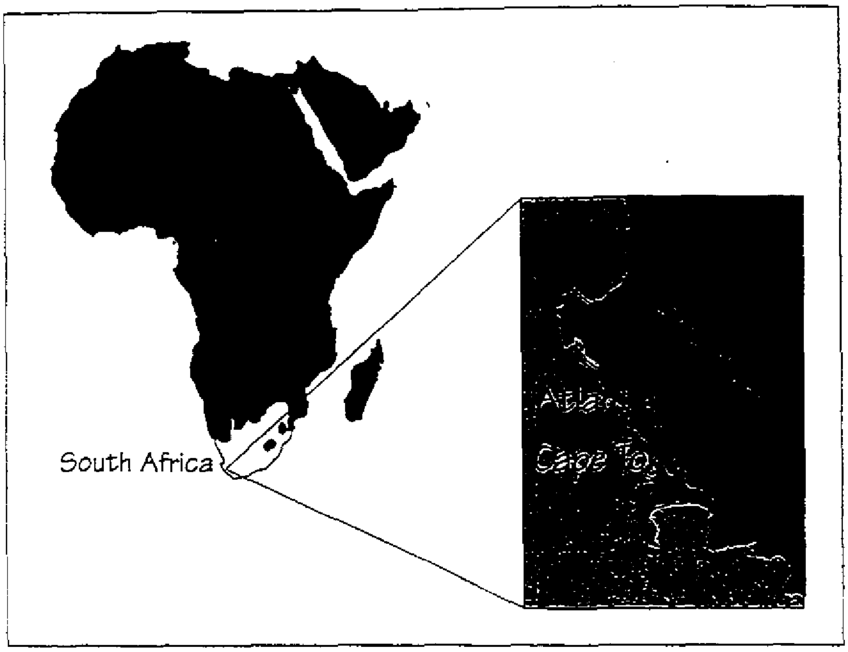

Figure 1 Location of Atlantis, South Africa

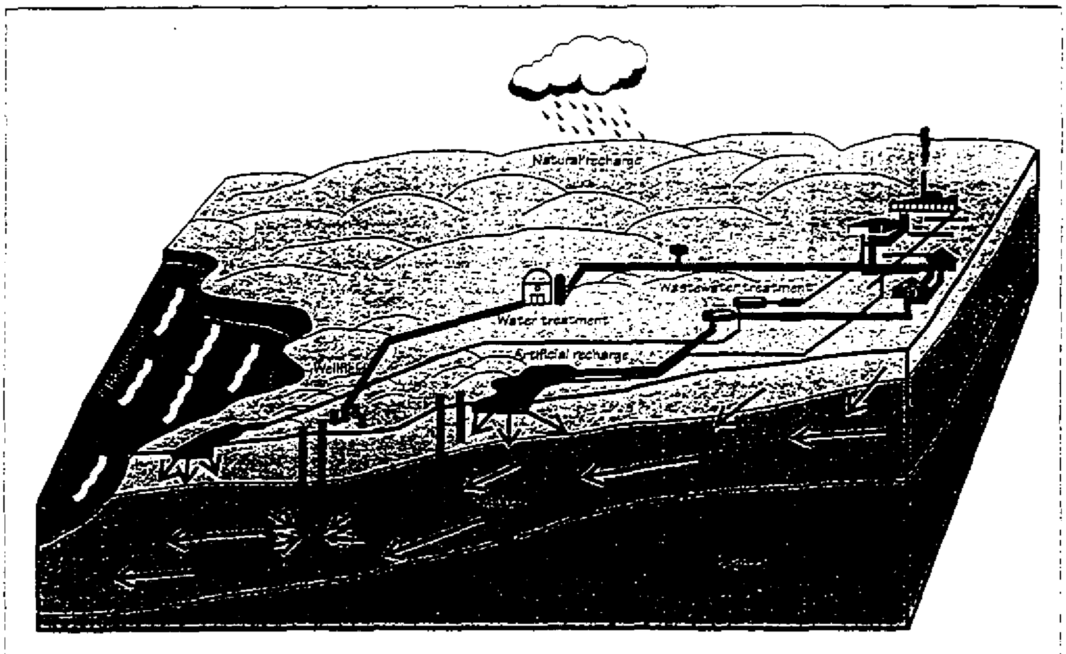

Figure 2 Three-dimensional representation of the Atlantis Water Resource Management Scheme

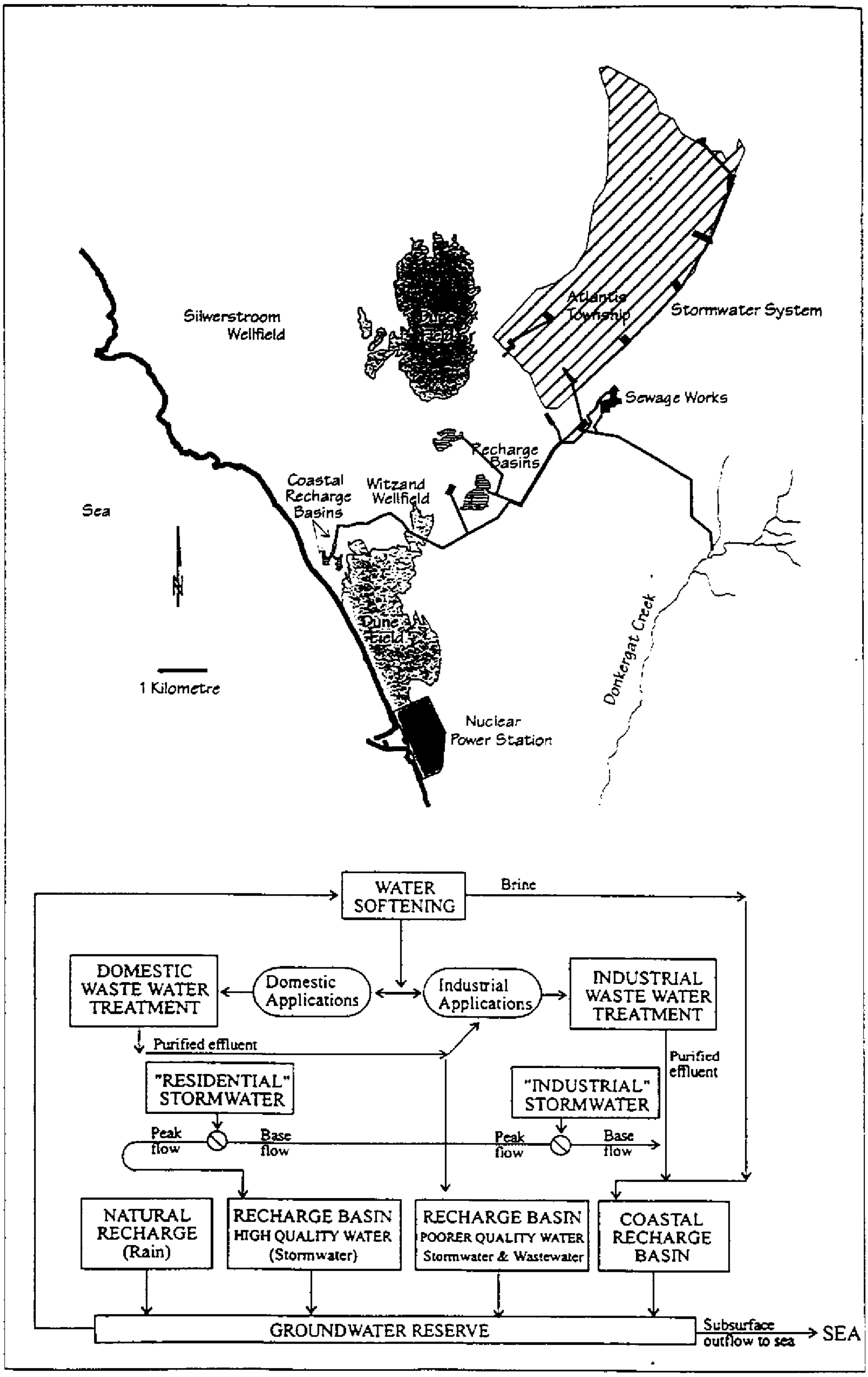

Figure 3 Atlantis Water Resource Management Scheme - Conceptual model and layout

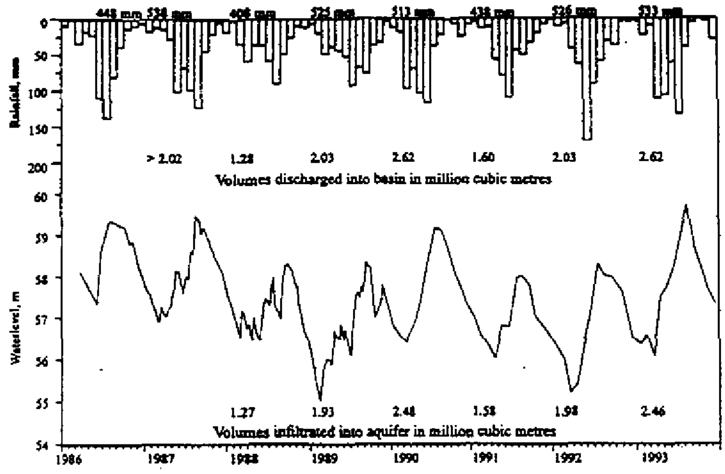

Figure 4 Annual volumetric totals for the poorer quality water recharge basin

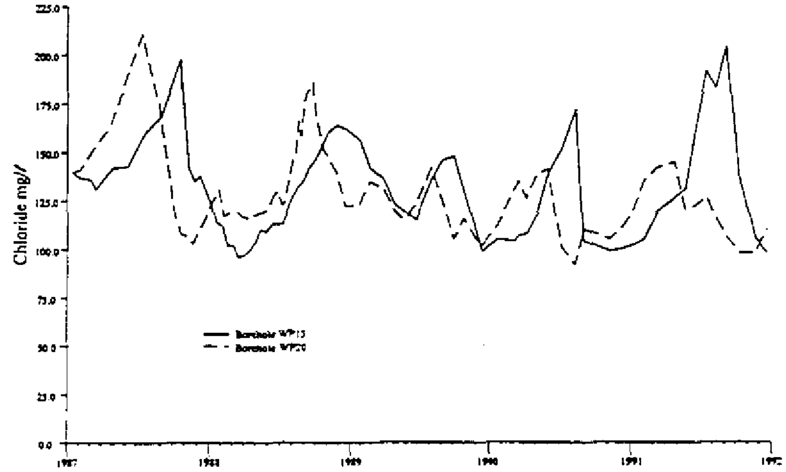

Figure 5 Groundwater quality trends as observed in two piezometers 80 m (WP20) and 200 m (WP15) down gradient of the recharge facility

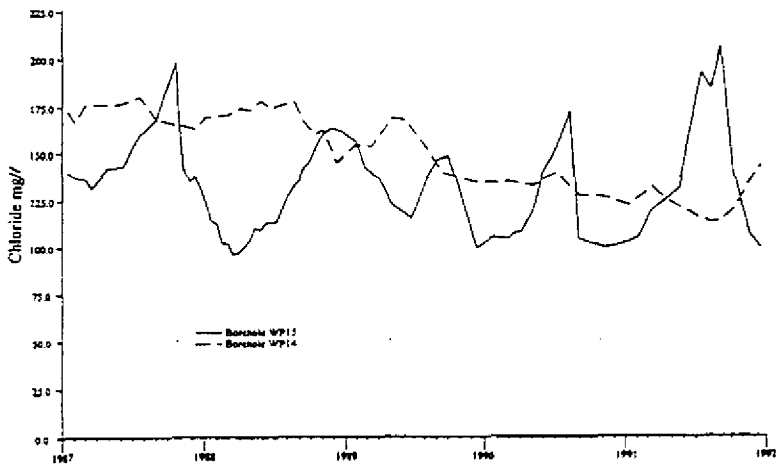

Figure 6 Groundwater quality trends in a pair of piezometers showing the different aquifer response at shallow (WP15) and deep (WP14) levels within the aquifer

GREYWATER RECYCLING IN PERTH AND KALGOORLIE

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ABSTRACT

Many householders in country areas of Western Australia are already recycling greywater. There is also an increasing demand for its usage in gardens in metropolitan Perth. Sources are washing machines, shower and bathwater. Evaporative air conditioner bleedwater is also used on the garden in country areas and this is being closely monitored. At present greywater recycling is illegal unless Health Department, Water Corporation and Local Government approvals are obtained for 'below ground' use. In some cases Dept of Environmental Protection approval is also required. Most of the applications are either illegal or are monitored as greywater recycling trials. As a water conservation measure, legal greywater recycling is being investigated by

- establishing trials under fully controlled conditions
- developing guidelines for Local Government application and administration
- trialing the guidelines at two Perth suburbs and in Kalgoorlie
- implementing through Health Act Regulation modification, the approval process at Local Government level with State government support.

The paper describes the process of monitoring the trials, developing the guidelines (adapting to the Urban Water Research Association of Australia (UWRAA) Guidelines from Brisbane), and setting up the trials for the guidelines. The individual site trials have been commenced in Geraldton, Cottesloe, Fremantle, Hovea and Palmyra.

KEYWORDS

greywater, recycling, drippers, garden watering, laundry waste, showers, bowl flushing, residential use, pilots, trial, guidelines, local government, park watering, demand management, wastewater reuse effects, health risks, sewerage impact.

BACKGROUND

Opportunities for wastewater reuse in Australia have been identified in previous publications (Schlafrig, Anderson, 1992) and reuse has only marginally increased since then. In the early settlement of Western Australia, sewage disposal was by means of a cesspit. This was no different from any other colonial settlement. With the introduction of the Health Act in 1911 the disposal of excreta by means of below ground cesspits was finally abolished. The first septic tanks were introduced to the State at the turn of the century and in 1927 the Health Act was amended to place control of septic tanks under the control of the Public Health Department. The first septic tank was a dual compartment single tank intended to treat only toilet waste. Other household wastewaters were disposed of separately. This had much to do with the toilet being an outhouse sited somewhere at the rear of the property.

Greywater is defined as untreated household wastewater which has not come into contact with toilet waste. It can be from bath, shower, basin, washing machine, laundry trough, kitchen sink and dishwasher. Currently kitchen wastewater is considered excluded and the bleed water from evaporative air coolers included for purposes of WA's reuse plans.

The opportunity to recycle greywater exists for other parts of Australia and overseas where schemewater must be conserved. Water, previously charged for on a rating basis, is progressively more subject to user-pays bases around the world, giving greater importance to viable reuse. Conventionally, household plumbing has conveyed all wastewater to either on-site treatment (septic tanks or other treatment units) or to sewerage systems where available. Water, as a flushing vehicle, has been from scheme or on-site supply.Under the health statutes ("*Treatment of Sewage and Disposal of Effluent and Liquid Waste Regulations*") the requirement is that all onsite wastewater be disposed of underground into systems such as soak wells and leach drains. It is also mandatory to connect into sewerage schemes where provided. More recently in Western Australia a view has developed that more use should be made of greywater. Water resources either in raw, treated or wastewater form in many country areas outside urban development are scarce and fully utilised (illegally) by separating the greywater from the blackwater and spreading it on gardens (lawn, flowerbeds or even the veges).

In Perth, where groundwater is more plentiful and of quality suitable for the garden there is less demand though, in some suburbs of country towns over 30% of householders are said to reuse greywater to some degree. There have been significant requests for approvals for greywater recycling and some trials are currently underway.

Brisbane City Council, through UWRAA has recently published "Model Guidelines for Domestic Greywater Reuse for Australia" (Jeppeson, 1996) and these have been used in developing draft guidelines for Western Australia. It is hoped that after trialing in a number of municipalities, the feedback will be sufficient to improve these guidelines and implement a formal process encouraging correct use of greywater recycling to ensure water conservation, public health risk minimisation and no significant detriment to the sewer system.

Fundamentally, the Health Department of WA (HDWA) is not averse to innovative methods of greywater recycling and supports the concept on the basis that systems;

- do not give rise to a nuisance or health risk

- are approved for the purpose and designed for long term use.
- are not considered an environmental risk

In 1995 the then Water Authority of Western Australia in '*A Water Supply Strategy for Perth and Mandurah to 2021*' listed as one of its commitments the preparation of guidelines and regulations along with the Health Department and Local Governments, for re-use of greywater

NATIONAL GUIDELINES

UWRAA Research Report

Guidelines for design and operation of greywater reuse systems have already been produced in Australia in both the national and local government spheres. In March 1996 the UWRAA, a division of the Water Services Association of Australia, published "Model Guidelines for Domestic Greywater Reuse for Australia" (Jeppeson, 1996). Jeppeson's proposed guidelines result, firstly, from a 1993 evaluation of overseas correspondence and literature, with some local greywater chemical and microbial analyses (Jeppeson, 1993). Secondly, Jeppeson in 1994, investigated overseas practices in greywater reuse and how they could be applied to Australia. Effluent application could then be as shown in Figures 1 and 2.

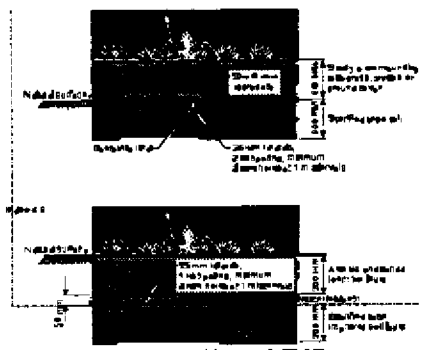

Figure 1 - Method 1,2 - Filtered Effluent

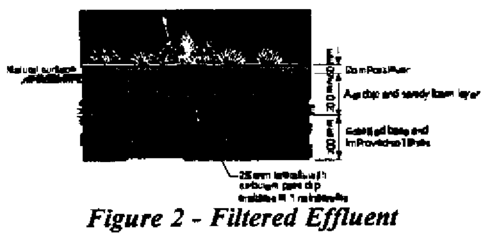

Figure 2 - Filtered Effluent

Interest in greywater reuse for water conservation at both federal and state government levels has prompted some local government bodies to offer advice to constituents. Examples are from Queensland (Dept of Primary Industries, 1992) and New South Wales (Recycled Water Coord Committee, 1995). Work has commenced in Western Australia on draft guidelines (guided by the UWRAA model) for use by local governments there (HDWA, 1996). It has been recognised that Local Government has a major role to play being the local administrator through Environmental Health Officers (EHO's) placed in each local government administration. If existing "informal" users or proposers of greywater systems were to lodge an application for approval having received a widely displayed set of guidelines (on trial), then the EHO could randomly check how these were complied with, any need for modification and extent of interest.

Scope

The Guidelines offer requirements for design and installation of new systems, as well as modifications for existing domestic greywater reuse systems. Coverage is given to;

- hand-basin toilets - automatic provision for handwashing, using this water for flushing the toilet
- primary systems - for direct reuse of untreated greywater from a single living unit and
- secondary systems - where short-term storage may be appropriate - includes multiple- occupancy dwellings

Audience

The model document aims to facilitate on-site reuse of greywater, carefully avoiding compromise to public health and the environment now and in the future. Clearly it targets local government environmental health administrators, who may then adapt it if appropriate, but there is excellent value for individuals who need cautious advice prior to seeking approval from local or state health authorities. As a significant first step to guide Australians in possible use of greywater, the UWRAA Guidelines will no doubt be the principal reference for all "would-be-reusers" nationwide. With overseas trends and a cautious approach they are a ready-made starting point for local governments whose experiences will hopefully be fed back later. In time, the WSAA may hopefully decide that an update is appropriate. Support is expected from the manufacturers of equipment for greywater recycling now that a base guide has been presented with which they can develop systems in liaison with authorities.

Method of Implementation of Guidelines

Initially, there will be a need for trialing in diverse (soil type) areas to acquire data for extension throughout the State. The steps involved will be finalising the draft guidelines, preparation of fliers for the ratepayers information, official launching in each area, monitoring of applications, random inspections of sites, reporting by municipalities and final reporting by the research team.

SITE TRIALS

In Western Australia a small number of trial greywater recycling installations exists. These have been authorised by the HDWA and the Water Authority, now Water Corporation, owned and funded either privately or publicly. They cover diverse conditions with testing in place for operational data and user information is also being gathered.

1. Geraldton

A three-year trial using all laundry and bathroom greywater from a single domestic house at Mt Tarcoola to irrigate ornamental garden areas. The project, funded jointly in January 1994 by the National Landcare Program and the then Water Authority of Western Australia, will help develop guidelines for small-scale use of greywater pretreated only for removal of hair and lint to facilitate pumping and small bore distribution. After screening, greywater is pumped to four distribution tanks (see Figure 3 below) and gravitates through small bore (10mm dia) distributors to sub-surface soak pots throughout garden areas. System monitoring starts with the householder keeping records of the types of principal additives, cleaning agents etc, discharged in the greywater. Analyses of the greywater-irrigated garden soil will be carried out for any residual effects on the sandy (with a small clay fraction) subsoil. Also a small hand basin for toilet users discharges to the toilet cistern. The principal test aspects are;

- screening and pumping raw greywater
- distribution by low-head gravity reticulation
- effect of greywater use on garden soil
- feasibility of using handbasin water for toilet flushing

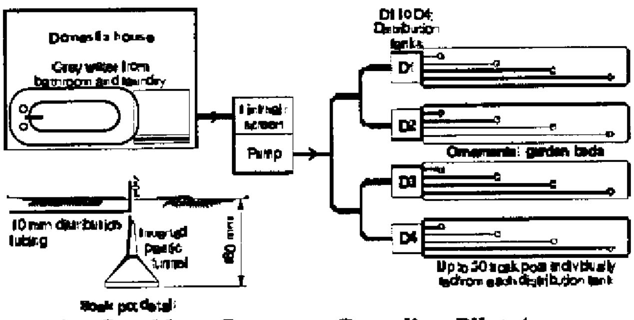

Figure 3 - Geraldton Greywater Recycling Pilot Arrangement.

2. Cottesloe

Greywater from the kitchen, bathroom and laundry of a single suburban home of four people is initially treated by a bioMax Model C10 anaerobic/aerobic wastewater treatment unit, modified for lower loading. Effluent, after chlorine disinfection, is pumped (see Figure 4. below) to underground Dripmaster drippers for irrigation of garden shrubs and groundcover. The installation is privately owned and operated. The principal test aspects are;

- effectiveness of modification to the bioMax unit for greywater treatment
- effectiveness of Dripmaster irrigation drippers

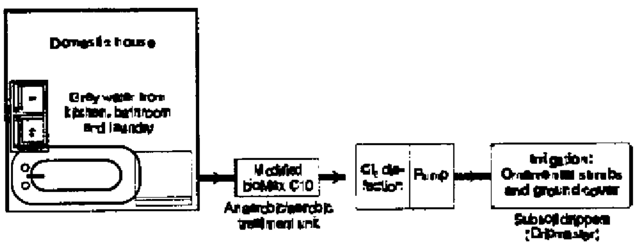

Figure 4 - Cottesloe Greywater Recycling Pilot Arrangement.

3. South Fremantle

Greywater from two homes is passed through a standard septic tank for sedimentation then used to irrigate shrubbery in a corner-block public park in built-up residential Fremantle.

The 1500 litre sedimentation tank meets HDWA septic tank standards and effluent can be diverted to either of two subsurface fields (see Figure 5 below). On one arm the effluent distributes onto an Ecomax-designed bed of red mud (by-product of bauxite to alumina processing) before takeup by vegetation. The other arm, a standby, distributes effluent along a subsoil drain laid in an organic humus filter bed. In neither case does greywater come to the ground surface. Surface and groundwater conditions are monitored. The settling tank, when examined in August 1996, had accumulated only a 20 to 30mm sludge layer since startup in March 1995. The principal test aspects are;

- effect of septic-tank settling on effluent quality and infiltration
- avoidance of surface ponding
- effect of irrigation of vegetation and
  effect on aquifer

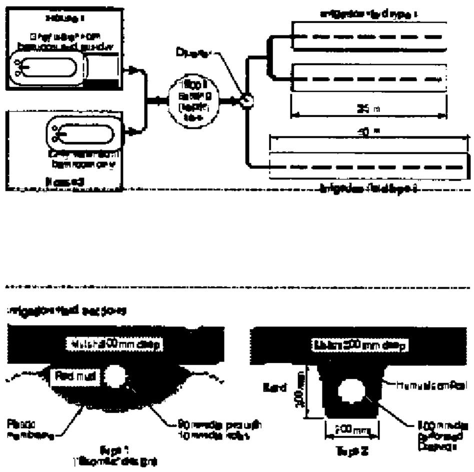

Figure 5 - South Fremantle Greywater Recycling Pilot Arrangement.

4. Palmyra

Greywater from the kitchens, bathrooms and laundries of six units for aged-couples, is biologically treated and disinfected for use in toilet flushing and garden irrigation as shown in Figure 6.

The project, funded under the Building Better Cities Program commenced operation in August 1995 and is owned by WA's housing authority, Homeswest. An Aquarius biological treatment unit produces a secondary-grade effluent for disinfection by 30-minute chlorine contact to not exceed "Guidelines for Use of Reclaimed Water in Australia" (NHMRC/AWRC, 1987), levels of thermo-tolerant coliform organisms. The effluent is stored for use for toilet flushing and irrigation of gardens. The units' gardens are fenced and ornamental only, situated on silica sand with a low clay fraction. The principal test aspects are;

- flow quantity measuring
- suitability of aerobic treatment unit for greywater (low organic load) treatment

- disinfection effectiveness after chlorination and after storage
- feasibility of reuse for toilet flushing and irrigation
- odour, regrowth and staining problems
- consumer acceptance

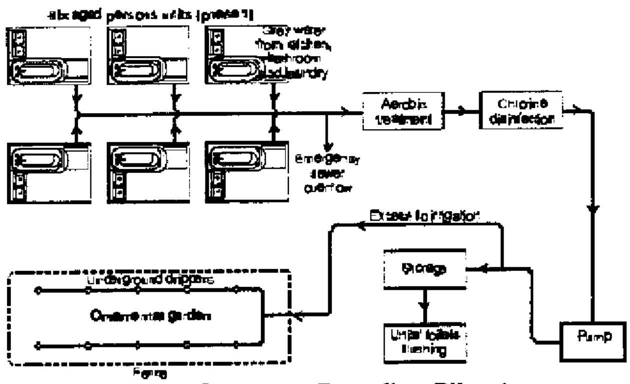

Figure 6 - Palmyra Greywater Recycling Pilot Arrangement.

5. Hovea

Greywater from bathroom and laundry of a four person home in the Perth Hills undergoes detention in a sedimentation tank similar to a septic tank meeting HDWA requirements. Effluent is distributed to a clayey soil by a sub-surface trench disposal system (see Figure 7 below) which complies with AS 1547 - 1994 "Disposal Systems for Effluent from Domestic Premises". The slotted drainpipes, surrounded by crushed stone, are laid in a parallel grid with a minimum clearance of one metre between them, fed on non-alternating basis in a garden environment. At 21 months since its January 1995 system startup the sedimentation tank had negligible sludge buildup and the soakage field functioned well.

The installation is permanent and its principal test aspects are;

- settlement as the sole greywater treatment
- effectiveness of the disposal/irrigation field in clay soil/garden environment

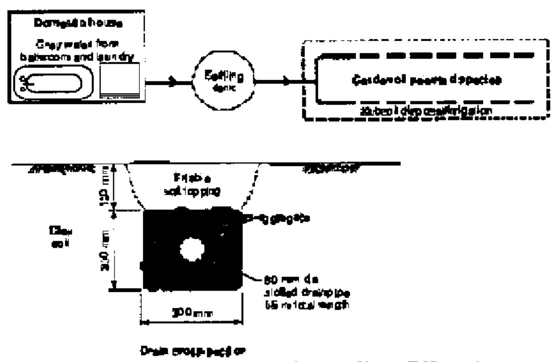

Figure 7 - Hovea Greywater Recycling Pilot Arrangment

PERTH SUBURBAN TRIAL

Appropriate suburbs or towns/cities needed to be identified which can be adapted for the purposes of the trial. The three broad areas of soil types for the whole trial are coarse sandy clay, rocky shallow base and fine silty clay. These occur;

- on the Perth floodplain (coarse sandy clay and sands)
- in the Hills (above the Darling Fault - rocky shallow base) and
- in the Goldfields (fine silty clay).

The Perth suburb of Bassendean is well known for its sandy soils and is representative of most councils inland along the coast but on the floodplain. This portion of Western Australia has a heavy dune and limestone belt running alongside the ocean in a strip up to three or four kilometres in from the coast. Kalamunda Shire, in the Darling Ranges, is quite different from Bassendean, having rock at very shallow depth and lateritic character in its topsoil, as well as clay/soil in the precincts below the foothills, allowing the opportunity to use this municipality for the second trial area.

Monitoring Setup

It is expected, as indicated earlier, that the WA Guidelines will be finalised in draft, issued to the appropriate officials in local government, advertised and a launch prepared and carried out. Fliers would be issued as a letterbox drop, application forms prepared and a database kept of all phone, mail, fax, inspections, etc. This will then be included in the report which would trigger the modification of Acts and Regulations.

Feedback

The main drivers for this research are the HDWA and the Water and Rivers Commission as the main interests are in public health protection and water conservation. Feedback from the public would be via the EHO's. During the trials, feedback will be encouraged but, apart from random checks, the process will be self-regulating and evaluation of this aspect will be part of the trials.

Evaluation

With the feedback being logged and penetration of the "marketplace" recorded, full evaluation of the trial will be carried out and appropriate recommendations made, including possible Regulation and Act modification.

KALGOORLIE TRIAL

Because of water conservation awareness and the implementation of the worlds first Waterwise City, Kalgoorlie-Boulder has registered strong interest and has been accepted for trialing the WA Greywater Recycling Guidelines and because of its different soil characteristics.

Soil Types
Soil types here include considerable clay and fine sandy silt, considerably different from Perth soils

Local Government Response
Communication was the main issue concerning the EHO at Kalgoorlie-Boulder City Council. A significant inquiry base or number would overload the central switchboard at the Council and special arrangements would be required along with the data recording and retrieval system for inquiries. Part-time assistance would be needed to relieve pressure from existing staff.

Community Support
The community consultation committee, WEAC, for the Kalgoorlie-Boulder Water Efficiency Program, expressed a clear wish to participate in the greywater trial and requested to be advised of progress.

HEALTH IMPLICATIONS

While limited microbiological examination of greywater has occurred in Perth, studies overseas indicate that greywater can be contaminated with pathogenic micro-organisms. Greywater, derived from human activity/contact, can contain pathogens, a health risk to humans if not handled properly.

Adequately treated wastewater can be applied above or below ground, however this does not mean that greywater application to the ground surface is considered acceptable. Apart from its health risk to humans further problems can arise when it pools and becomes stagnant. Breeding of mosquitoes, attraction to other insects and the nuisance from odours are all potential problems. Greywater would have to be treated including disinfection for any surface disposal.

Disposal of greywater below ground is the preferred option. Provided the greywater is discharged at an adequate depth below the ground to prevent surfacing this, it is reckoned, provides adequate protection to public health. The only form of treatment that will be required would be settlement to reduce suspended solids, oil, greases, etc.

IMPLEMENTATION IN WA

Jeppeson (1994), suggests that new guidelines (Jeppeson, 1996) be read in conjunction with the bye-laws and regulations of relevant regulatory and/or administrative authorities. National implementation of greywater guidelines, though desirable, would need time, experience and education with the use of numerous systems before a comprehensive set was adopted. Probably the biggest issue for local implementation is the local government ability to support the approval and random inspection process, together with the readiness of existing greywater recyclers to obtain approval.

For different soil types, there will be different performances. The Guidelines will need to address the diversity of geotechnical conditions in residential properties throughout WA.

Regulations

After feedback from trials and extension to other areas, review of Acts and Regulations can be carried out. Changes to some regulations may be necessary for wider use of greywater in the community. Health, water and environmental rules aim to protect public health and the environment from the types of problems resulting from indiscriminate discharge of community wastewaters. Carefully considered results from carefully planned trials will enable liaison between authorities and other stakeholders to produce optimum safeguards while utilising greywater for significant saving in water. So, an emphasis would be placed on use, not only disposal, accounting for

- avoidance of surface ponding or runoff
- control of type of use of and access to irrigated areas and
- avoidance of contamination of surface and underground water
- prevention of deterioration or nutrient overloading of the soil

Some flexible aspects of the laws, primarily the health legislation need to allow for further findings of trials and effects of local conditions

Communication

Already an essential liaison exists within the "Greywater Recycling Policy Group" between Health Department of WA, Water and Rivers Commission, Department of Environment Protection and the Water Corporation, whereby existing trials are monitored and proposals are evaluated to assist the determining authorities where necessary. Innovation, an essential element for cost-effective systems, will no doubt spring from the private manufacturer and customer (as it has done for other household watersavers). A "central" body representing at least those of the above group must exist for;

- monitoring new proposed innovations and existing trials in WA
- maintaining a cautious outlook on greywater reuse, realising its potential hazards
- keeping constant contact with other states and leading overseas "users" of the technology
- recommending possible amendments to the regulations
- ensuring that in administration of the rules, the basic intent to protect both the public health and the environment is effective.

Training

Not only must householders be given informative guidelines to minimise the chance of poor greywater practices but the industry which might provide these guidelines, must itself receive effective training from Local Government in;

- awareness of the benefits, water saving, nutrient value etc of greywater use for irrigation
- current regulations and the principal hazards they are meant to address
- operation of systems especially those serving more than just single dwellings
- basic precautions during design and installation eg limiting access to some areas, avoiding cross connections with other fluids, site and soil selections etc
- sensing and dealing with problems arising eg hydraulic overloading chemical intervention, soil clogging etc

A principle on which training might be based is "simplicity is the key, safety is a necessity".

EFFECTS ON SEWERAGE

Any household which reuses greywater would reduce its wastewater effluent by around sixty percent leaving toilet to sewer or septic system. (It is possible to also utilise kitchen sink rinse-bowl effluent as greywater). The designers and operators of wastewater systems have an interest in the aesthetic and financial effect that the reduction may have on sewers, treatment plants and effluent disposal systems. The benefits of saving potable water are complemented by having less wastewater to transport, treat and dispose of for most of each year, ie outside the wet period when greywater would normally revert to sewers and septic systems. Any significant problems for sewers etc, due to a summer flow reduction, must be addressed.

Sewer Flows

Jeppeson (1993) estimates an annual saving in wastewater discharges in Brisbane to be 105kL per household (356 L/d). Wastewater pumping savings will accrue depending on the sewerage system layout. Perth, for example, has an unusually high number of pumping stations, some operating in series, and could reduce energy and maintenance costs significantly with on-site reuse of greywater outside the winter period. Concern with possible sewer blocking resulting from reduced flows could be investigated in-field, in a similar manner to the site testing carried out (in at least three states of Australia in 1990) to examine the effects of reduced WC flushing quantities on private and authority sewers laid at minimum gradients. Possible effect on sewer septicity will be assessed.

Wastewater Treatment

It has been estimated (Jeppeson, 1993) that each average household could reduce both its biochemical oxygen demand (BOD) and suspended solids (SS) output by 10.5kg over the summer greywater reuse period. This assumes greywater BOD and SS concentrations both to be 100mg/L. Total loadings on both primary and secondary treatment plants being reduced may affect process parameters in the longer term. An additional effect may be noticed from movement between on-site and sewer during the change to wet weather greywater discharge. Aeration treatment systems could gain through air-use reduction. Biosolids treatment and disposal must also benefit in the longer-term. Possible reduction in biosolids byproducts (methane, oil, soil conditioner) may not be sufficient to classify as a loss.

Effluent Disposal

Where effluent is pumped away (eg Subiaco and Woodman Point) some benefit from energy-saving will occur, even if not obvious in the short term. For smaller schemes, especially in summer, a reduction in effluent quantity will assist in such matters as visual impact and eutrophication in disposal areas, river beds etc. Such issues are not so sensitive in winter time. Some country town schemes however have all their treated effluent reused for irrigation so greywater reuse would probably not benefit their scheme effluent disposal process.

For on-site septic tank system users greywater reuse in the summer period can be the means of extending the life of their soil absorption system (soak-well, leach drain, french drain, ET bed, etc) through reduction of its load (by 60% hydraulically) for around eighty percent of each year. It is interesting that for Brisbane with 275000 water connected households, the estimation by Jeppeson (1994) for dollar savings if all customers were to install greywater reuse systems is;

- for potable water saving, \$19m to \$31m per year
- for sewerage system saving, \$11m per year (Brisbane assumed to be 98% sewered)

Economics of greywater recycling systems and its beneficiation will depend on the extent of recycling, water and sewerage cost and details of the system.

CONCLUSIONS

Widespread informal and illegal greywater recycling needs to be adapted in the community interest to guidelines which enable the consumer to utilise the resource whilst minimising health risks. Proposed trials may enable the refinement of guidelines and procedures suitable for use in Western Australia, allowing local government to participate more in water conservation, a topic fresh in the minds of Western Australian water consumers. Cost beneficiation can be carried out but economic benefit to the consumer will vary, with most not attractive until environmental cost benefit is considered with system benefits distributed over rates. The economic perspective for the utility (water and sewerage) is more favourable but this would require validation by trialing and monitoring quantitative and qualitative impacts on all aspects of sewerage.

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LOW COST SYSTEMS

EFFECTIVENESS AND USER ACCEPTANCE OF COMPOSTING TOILET TECHNOLOGY IN LISMORE, NSW.

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Abstract

Composting toilets use no water and have the potential to remove much of the pollutant load from the aqueous waste stream of the average Australian dwelling. In New South Wales a number of commercially available units have been approved for use in unsewered areas, and guidelines for owner-built units have recently been released. While there is growing confidence in relation to the environmental and public health credentials of this emerging technology, concerns still linger with regard to issues such as ease of management and acceptance by the general public.

A survey was conducted on composting toilet owners in the Lismore City Council area (northern NSW). A total of 89 surveys were mailed out and responses were received from owners of 24 owner-built and 5 manufactured units representing eight different toilet designs. Roughly half of the respondents own continuous flow systems while half own batch type systems. Fifteen of these respondents were chosen for later site visits.

A number of conclusions are drawn in relation to the strengths and weaknesses of owner-built vs. manufactured units; large vs. small chambered units; and batch vs. continuous flow configurations. Ninety-three percent of respondents rated the performance of their units as "very good" or "excellent" and have experienced few problems. The other 7% returned a verdict of satisfactory and no units were rated as poor. Because of their adaptability and passive ventilation systems the large chambered, owner built Minimus and Farralones Batch designs show most promise of meeting the objectives of "minimum maintenance and supervision" required "to meet the challenge of small scale decentralised treatment" which are the focus of this conference. Any attempt to transfer composting toilet technology to developing countries would have to be done on a community scale and be integrated with social and cultural development.

Keywords

Composting toilets, continuous flow, batch system, on-site sewage, performance, user acceptance.

The Modern Composting Toilet

Van der Ryn and Cowan (1996) report that the first modern composting toilet, the Clivus Multrum, was invented by a Swedish engineer for use in summer cabins built on thin glacial soils adjacent to mountain lakes. Such situations were totally unsuitable for standard onsite wastewater systems incorporating septic tank and leaching field. Figure 1 shows the essential features of a modern adaptation of the original Multrum design. Human faeces and urine drop into the chamber via the stool while kitchen scraps and other organic waste are added from the garbage shoot (*sic*). Carbonaceous bulking material can be added as necessary via the stool to adjust moisture content and carbon to nitrogen ratio. The material moves slowly from left to right down the sloping floor. Aerobic conditions are maintained in the heap by air which enters through an intake adjacent to the access door. In modern versions of the Multrum the active ventilation system is driven by an electric fan in the exhaust duct. A drain at the low point of the chamber (not shown) removes excess fluid. The chamber is made from high density polypropylene with fibreglass baffles. The Multrum is an example of the generic "continuous flow" configuration, so called because once commenced, the composting process continues for as long as material is added. Periodic removal of compost occurs as necessary through the access door.

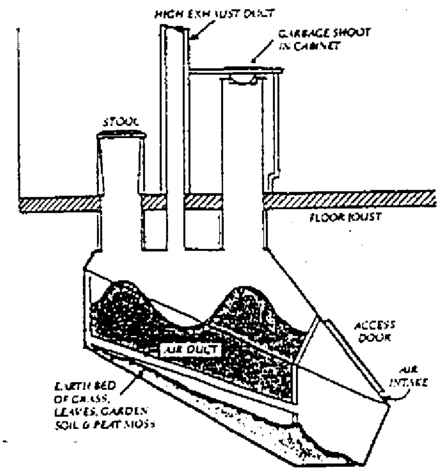

Figure 1: Cross section of a Clivus Multrum
Source: Clivus Multrum Australia (undated)

The other generic composting toilet configuration is the batch design. Van der Ryn and Cowan (1996) suggest that the first American batch systems were developed in California during the 1970's by homesteaders who found the Clivus Multrum too large and expensive for their small homes and even smaller budgets. Based on the design of the standard concrete block composting bay promoted in organic gardening books of the day, the Farralones Batch design became popular with do-it-yourself rural homesteaders in both the US and Australia. Figure 2 shows the two chambers with the vent pipe which drives the induced draft passive ventilation system. Faeces, urine and any compostable material are added through the squat plate, a hole in the floor which may be enhanced by a pedestal if the users favour the standard European seated position for defecation. The composting material is suspended above the chamber base on a permeable false floor (usually galvanised steel mesh) to facilitate flow of air

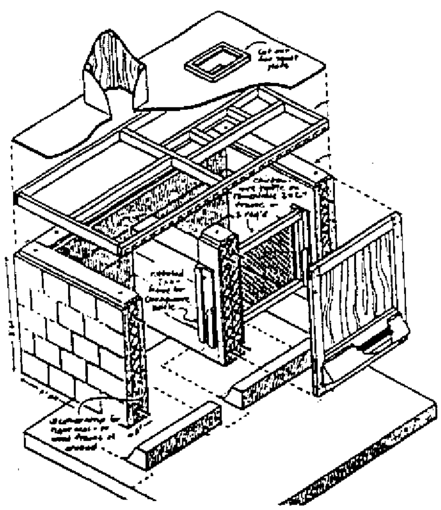

Figure 2: The Farralones Batch System
Source: Van der Ryn, 1980

vertically up through the heap. When the chamber is full it is closed off and allowed to stand "fallow" while the second chamber is opened for contributions. When chamber No. 2 is full, the first batch is removed from Chamber No. 1 which once again becomes active . . . and so the cycle proceeds.

Composting Toilets in the Lismore City Council Area

Since the early seventies the Lismore area in the north eastern corner of New South Wales has been the centre of a "back to the land" movement which continues to this day. For many of the "new settlers" who moved onto the cheap degraded farmland made available by the rural depression of the time, the desire to "live lightly on the Earth" was a commonly held value. Despite generally high rainfall, the region experiences a long dry spring and many new settlers establishing homesteads on dry ridges could not see the sense in using scarce water to shift their excreta the few metres from the privy to a septic tank. So began a search for more rational excreta management technologies in the area. The first Farralones units were built in the late seventies and in the early eighties the first Minimus (a concrete block adaptation of the Clivus Multrum - see below) appeared. Human ingenuity, being what it is, these designs were soon varied and adapted by local tradesmen and owner builders to suit their own needs. By the mid eighties Lismore City Council, despite early fears about odour and human health, and concerns about the fact that the owner-built units were not approved by State Department of Health, had begun to adopt a positive attitude towards these experiments. Kohlenberg (1996) suggests that Council had the choice of (i) turning a blind eye to the new unapproved technology, (ii) prosecuting the "offenders" or (iii) taking the opportunity to monitor and evaluate the new systems. Council chose to adopt option (iii) to grant approval (in principle) for owner built units on a trial basis.

At the same time as these early trials were indicating that well managed composting toilets did not produce the predicted offensive odours and threats to human health, a number of studies were showing that the officially sanctioned water-based septic tank technology was itself very fallible. Indeed a study by Geary (1992) on the village of Nimbin demonstrated widespread failure of septic systems in the Lismore Council area itself. A study by Safton (1993) on six composting toilets in the Lismore region (four of them owner built) concluded that "the systems are in fact working with respect to the destruction of parasites and commensals." This was despite the fact that the composting chamber temperatures never rose significantly above ambient. Safton concluded that "the humus/end product could therefore, with the exception of viruses, be considered pathogen free." In 1997 the NSW Health Department (1997) issued guidelines for the approval of composting toilets (including owner built units).

As time passes and as experience with composting toilets expands and deepens, it is becoming obvious that the technology has much to offer a society seeking ecologically sustainable solutions to problems of resource depletion and environmental pollution. Questions arise, however, in relation to issues such as ease of management and general public acceptance.

Objectives of the study

The objectives of this study were:

- to determine, by means of written survey and on-site inspection, the performance and ease of management of composting toilets in the Lismore City Council Area;
- to evaluate the level of user satisfaction with the systems currently in use; and
- to estimate the strengths and weaknesses of the various designs in use with a view:
  - to making recommendations to prospective composting toilet owners, and
    to facilitating the evolution of the technology.

Mail Survey

The Lismore Local Government Area covers 1,267 km^2 and is home to 45,000 people, of whom some 16,000 are not served by a centralised sewage system. Climate in the study area is sub tropical with a mean daily maximum temperature of 31.8°C in mid summer and 20.5 °C in mid winter. At the time of the survey (June 1997) Lismore City Council had approved 89 applications for composting toilets. By the end of October this figure had grown to 108. Table 1 shows the number of annual approvals for composting toilets, aerated systems and standard septic systems from March 1991 to October 1997. It is worth noting that the number of approved compost toilets as a proportion of total on-site system approvals has grown steadily from 2% in 1991 to 22% in 1997. Council estimates that the number of non approved systems would be at least equal to the number of approved systems..

Table 1: Number of approvals per year for composting toilets and other on-site systems in Lismore City Council Area. *Source: Lismore City Council*

year	1991	1992	1993	1994	1995	1996	1997 to 30/10	total
compost toilets	3	11	7	18	20	26	23	108*
aerated systems	9	17	36	44	35	22	31	194
septic systems	116	104	101	107	82	78	53	641
total	128	132	144	169	137	126	107	943
compost as % of total	2	8	5	11	15	21	22	

* at the time of survey (June 1997) this figure was 89

Surveys were mailed to all successful applicants in June 1997. The survey sought information on:

- design configuration (continuous flow or batch);
- method of manufacture (owner built or manufactured);
- loading rate (frequency of use, frequency of peak loading);
- bulking material (type and frequency of addition);
- performance (*e.g.* occurrence of odours, excess moisture, design or operator problems);
- maintenance operations (*e.g.* raking heap, removing end product);
- management and use of end product;

- degree of satisfaction with the unit; and
- any suggested design modifications based on experience so far.

Of the 89 surveys sent 29 were returned completed. A further 15 were returned incomplete by people who had either not yet installed their unit or not used it long enough to make the necessary evaluations. Table 2 summarises some of the basic features of the eight toilet designs covered by the responses, along with a summary of user performance ratings for each design.

From Table 2 it can be seen that of the 29 responses only 5 (17% of total) were from owners of manufactured units. This proportion is below the 25-30% estimated by Council staff for manufactured composting toilets in the Council Area. The high proportion of owner built units is perhaps a reflection of both the strong do-it-yourself ethic in rural parts of the region and the economic circumstances of many rural residents. While the price of manufactured units ranges up from $2,500, it is claimed that materials for the simplest owner built unit (the Pickle Barrel) can be obtained for as little as $100. Materials for the Minimus cost about $500 and a local tradesman has been charging an additional $700 for labour.

All of the units surveyed were rated as *satisfactory* or better, and 93% (27 units out of 29) received a rating of *very good* or *excellent*. Because of the extremely small sample sizes it is impossible to draw any definitive conclusions on user performance rating for the manufactured systems. However the three major owner built designs all averaged a user performance rating between very good and excellent. There were more than twice as many responses from Minimus owners as for the next highest design (the Farralones Batch).

Site Inspections

Of the 29 respondents to the mail survey, 27 indicated a willingness to participate in a site inspection by one or more of the authors. Twenty of these were inspected on 15/8/97 and 31/8/97. The number of on-site visits per design is noted in column 3 of Table 2. The site visit included an unstructured interview with the owner on matters relating to perceived performance, management and bulking materials. Each unit was inspected with particular attention paid to odour, condition of end product and presence of excess moisture.

Results

The results of the mail survey and site visits for each of the eight designs are summarised by Pollard (1997). What follows here is a distillation of that summary.

<u>Why choose the composting toilet?</u>
The main reasons given by respondents for choosing a composting toilet in preference to other systems were:

- to save water;
- to create a resource (*i.e.* compost);
- to avoid polluting waterways;
- to avoid environmental problems as site not suitable for septic system;
- to save money as this option appears to be relatively cheap to install and maintain;
- to save time as this option appears to be easier to maintain than alternatives;
- to take responsibility for one's own bodily excretions; and

- to create a multiple use recycling facility for scraps, garden waste and paper as well as excreta.

Table 2: Summary of characteristics and user performance rating for the eight composting toilet designs.

System	Response rate	No. of insp'n onsite	Owner-built or Manufactured	Size *	config-uration **	Ventilation Active (A) Passive (P)	User Performance Rating			
							excell	v. good	satisfac	poor
Clivus Multrum	2	1	Manufactured	L	CF	A	1	1		
Rotaloo	1	1	Manufactured	S	B	A			1	
Natureloo	1	1	Manufactured	S	B	A		1		
Dowmus Dry	1	1	Manufactured	L	CF	A		1		
Farralones	6	2	Owner built	L	B	P	3	2	1	
Minimus	13	11	Owner built	L	CF	P	7	5	1	
Wheelie Batch	4	2	Owner built	S	B	A	1	3		
Pickle Barrel Batch	1	1	Owner built	S	B	A		1		

* Size of chamber: L = Large (>1,000 L) , S = Small (<500L)

** Configuration: B = Batch, CF = Continuous Flow

Major management issues for the eight system types surveyed are now dealt with. Where appropriate a brief description of the system is also given.

Clivus Multrum (See Fig. 1 above)

Neither of the two units surveyed is subjected to heavy peak loadings. Both units receive kitchen scraps. Wood shavings and sawdust are used as bulking material. One unit experiences odours if the fan is not operating or if the pile had not been raked recently. Neither unit experiences a problem with excess liquid which is directed into the greywater system in one case and into an absorption trench in the other. In both cases the top of the heap is knocked off every 2 months, raking occurs every 6 months and cleaning the chute occurs every 2 weeks. Compost from the inspected unit was odour free, dry and friable and was used around native trees and shrubs. User performance ratings were *very good* and *excellent* with the only major problem being the presence of vinegar flies introduced through improperly stored fruit and vegetable scraps.

Rota-loo

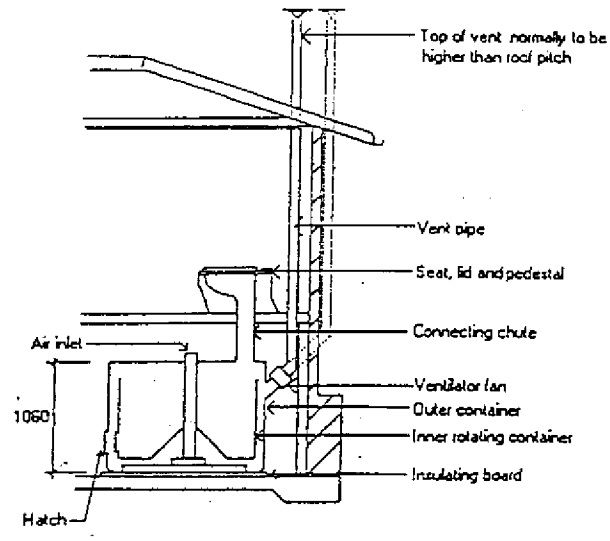

Figure 3: Cross section of a Rota-loo
Source: Environment Equipment (undated)

Recent versions of this batch system (Fig. 3) incorporate 6 composting containers mounted on a central spindle. When a container is full, the assembly is rotated through 60° to align the next container with the pedestal. Liquid drains into a tray and evaporation is facilitated by a 250 watt immersion heater and fan powered active ventilation. Excess liquid can be drained off. The unit surveyed in this study was an older model incorporating only 2 containers. Installed in a small factory it gets 10 to 15 uses per day mainly for urination. There are regular heavy peak loadings which do not adversely affect performance. Sawdust is added once per day. The chute is cleaned weekly, and the top is knocked off the pile once per month. Material is removed every 3 to 5 months and is placed in a lidded container for further breakdown prior to being added to a vegetable garden. The one Rotaloo surveyed was rated as *satisfactory* by its owners.

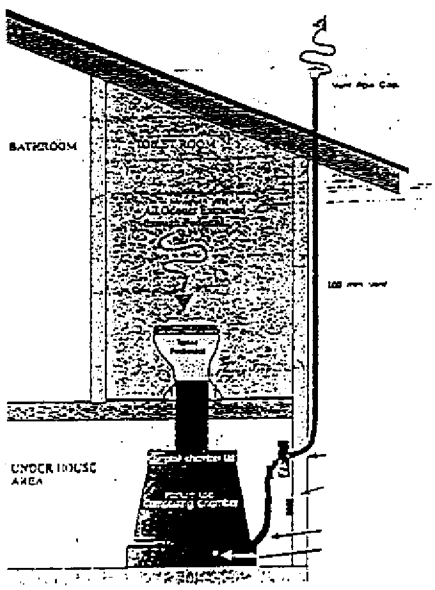

Figure 4: Cross section of a Nature-loo
Source: Nature-loo (undated)

Nature-loo

The chamber of this manufactured batch system (Fig. 4) is horizontally divided into a 150 L upper composting compartment and a lower 85 L evaporation section. Active ventilation creates sufficient evaporation to remove liquids in all but peak loading situations. At such times excess liquid is removed by a 19mm pipe. When full, the chamber is removed from below the pedestal where it can be connected to the ventilation system to facilitate aeration during the "fallow" stage. The one Nature-loo surveyed did not take advantage of this feature, without detriment to the composting process. This unit has been in use for 10 months and receives up to 10 visits per day from the two residents. It is also subject to periodic heavy peak loadings. These do not create noticeable moisture or odour problems for the owners. Sawdust and kitchen scraps are added once per day. The (solar powered) fan is turned off at night to save power and this is probably the cause of odour in the morning. One problem has been the failure of two fan motors, presumably due to moisture and dust from bulking material in the duct. Solution - don't use fine sawdust in actively ventilated systems. User performance rating for this one unit was *very good.*

Dowmus Dry
This manufactured continuous flow system uses worms to break down human excreta and other compostable household refuse. Variations on the configuration shown in Fig. 5 enable the polypropylene tank to be placed beneath concrete slab floors because final product is lifted out by an auger. The unit surveyed averaged 10 uses per day and had not been adversely affected by heavy peak loadings in the 2 years since installation. In this time it had not been necessary to perform any maintenance operations at all. Sawdust and kitchen scraps are added once per day. The owners rate their Dowmus as *very good.*

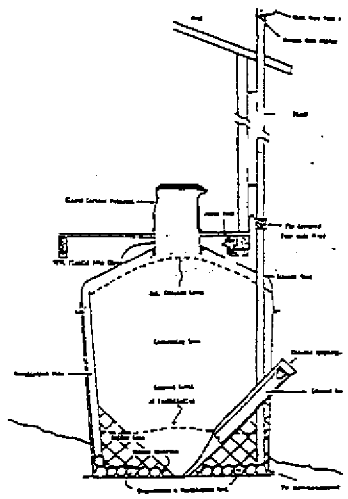

Figure 5: Cross section of a Dowmus system
Source: Dowmus (undated)

Farralones Batch (see Fig. 2 above)
One of the more common owner built designs in the Lismore area, this passively ventilated batch system can be made from concrete blocks, rocks or any locally available material. This versatility makes it suitable for service in developing countries. The standard unit has 2 chambers, each with a volume of slightly less than 1,000 L. The dimensions can be varied to suit available space. Materials for the six units surveyed included brick, mud brick, fibro and steel, concrete block, and rock and mortar. The units were used regularly by 1 (2 cases), 2 (2 cases) and 4 (2 cases) people. Three of the six units received regular peak loadings without problems. All 6 users added bulking material after each use. These materials included sawdust, wood shavings, garden waste and shredded paper. Three units received kitchen scraps without problem, while one owner attributed problems with vinegar flies and excess moisture to this practice. Five of the units had no liquid drain and experienced no problems with excess moisture. Average residence time for the six units varied from 12 to 18 months, and the compost was used to fertilise fruit trees, bananas, flower gardens and shrubs. None of the Farralones owners used the end product on vegetable gardens. The major maintenance task with this design is associated with start-up of a new heap. Most owners prime the new heap with a mixture of bulking material and garden soil. Excess moisture can be a problem in the early stages of a heap's life until a sufficient volume of material accumulates to retain the added moisture. A design problem that came to light during the survey relates to the fact that the front access panel must be removed to inspect the heap or to remove final product. Unfortunately this often causes material to fall out of the chamber. Despite this the average user acceptance rating for the Farralones units was between *very good* and *excellent.*

Minimus
The Minimus is a low cost passively ventilated adaptation of the Clivus Multrum design. The first units, installed in Canada and The Philippines in the early seventies were made from concrete blocks and PVC pipes. The first Australian Minimus, was built in the Lismore area in 1984. It was also the first Council approved (in principle) owner built composting toilet of

any description in the Lismore Council area and probably the first in Australia. Excreta and bulking material drop through the hole in the squat plate (Fig. 6) to form the beginnings of the pile. The growing pile is aerated by the four ducts which channel air into the heart of the heap. The 30° slope of the floor causes slow movement of the heap from the compost chamber on the left to the humus chamber from which it is removed. The standard Minimus has a capacity of approximately 2,000 L which ensures a long residence time. The chamber height of 2 m tends to restrict this design to hilly sites or to houses with high floors. The fact that over 40% of respondents to the survey were Minimus owners is an indication of the popularity of this design in the hilly country to the north of Lismore. Over the years the basic Minimus design has been adapted to suit the needs of householders. Dimensions have been changed to accommodate under-floor space availability, and various construction materials have been used (Davison, 1996).

The average number of regular users of the 13 Minimuses surveyed was 3. Nine of the units are subjected to heavy peak loadings and some odour and increase in moisture levels was reported under these conditions. In each case this was attributed to lack of visitor experience and inadequate addition of bulking material. Three of the units are located at busy sites (a school, a shop and a communal residence) and are used frequently. One system is regularly used by at least thirty people per day. Eight of the owners add bulking material after each use while two add it once per day. The remaining three users vary the interval between additions from daily to weekly without resulting odour problems. Materials added included sawdust, wood shavings, shredded paper, kitchen scraps and lawn clippings. Only seven of the units are currently fitted with a liquid drain. Maintenance operations were carried out every one to two months and included knocking the top off the heap and cleaning the chute. Minimuses in private homes appeared to be better maintained than those in communal spaces. The majority of Minimus owners gave a performance rating of *excellent*.

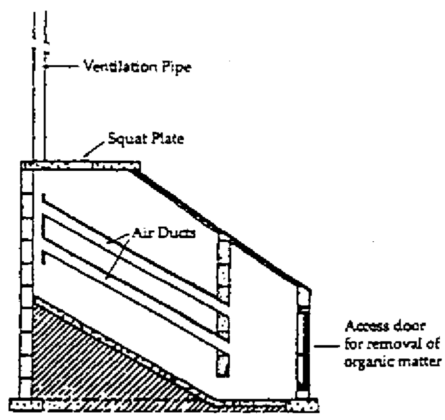

Figure 6: Cross section of Minimus
Source: Hupping-Stoner (1977)

Wheelie Batch

Designed by Lismore based consultant Stuart White, the Wheelie Batch toilet started out as an owner built version of commercial batch systems like the Nature-loo. It is now available in kit form and as a fully assembled item. The chambers are 240 L mobile garbage bins (wheelie bins) modified by the addition of a false floor and interior ventilation system (Fig. 7) made from 3 vertical 100 mm PVC pipes. Like all small chambered units it requires active ventilation to achieve aeration and moisture reduction. As with the Nature-loo the fallow bin is connected to the ventilation system to maintain aerobic conditions and enhance breakdown. Early problems with blocked drain holes have been eliminated by upgrading to a larger hole. All four users reported slight odours after peak loadings or periods with the fan turned off.

These had been overcome by addition of bulking material and turning on the fan. Wheelie Batch owners rated their systems as either *very good* or *excellent*. The main complaint was the short "fill-up time" (3 months for one unit serving a family of three). In order to comply with the Guidelines' (NSW Health , 1997) requirement of a minimum residence time of 12 months this system would, in theory, need 4 bins (3 fallow and 1 active).

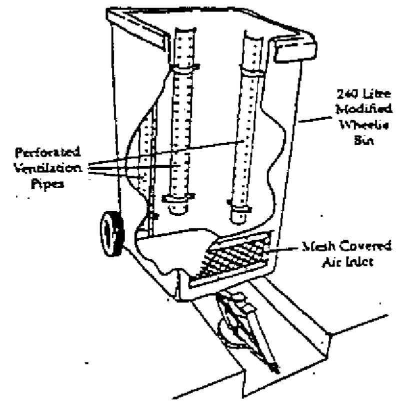

Fig 7: Cutaway view of Wheelie Batch
Source: White (undated)

Pickle Barrel Batch

This owner built, actively ventilated batch system has much in common with the Wheelie Batch. The major difference is that the 240 L garbage bin is replaced by a 200 L polypropylene pickle barrel fitted with a false floor 30 cm from the bottom of the barrel. Aeration is achieved by the attachment of 4 vertical perforated 100 mm PVC pipes to the inside of the barrel wall. This design is probably the cheapest option for the owner builder. The one unit surveyed has received an average of 10 to 15 visits per day and has been subjected to periodic peak loadings during its one year of operation. Although a fan is fitted the owners report that they rarely need to turn it on. The original bulking material of straw was felt to inhibit rapid breakdown (too coarse). Medium grain sawdust is currently being used. No kitchen scraps are added. Two people take 8 months to fill the bin.

Discussion

Management issues - design related and behavioural

The two major types of management issue to emerge from the survey are "design related" and "behavioural".

Examples of design related issues are the modification of inadequate drainage and ventilation systems. A behavioural issue that came up for some respondents was the type and frequency of addition of bulking material. In the case of the Nature-loo which experienced two fan failures in 10 months a design problem (inadequately protected fan motor) was overcome by a behavioural modification (stop using fine sawdust). It appears that where problems have occurred owners have been able to adapt their behaviour to overcome them. It is also apparent that, particularly with the owner built designs, an ongoing process of modification and adaptation is occurring.

Choosing a design for a particular application

Composting toilets come in a variety of shapes and sizes and in a number of price brackets. Clearly each design has its strengths and weaknesses. For the purpose of analysis a three dimensional approach is taken. The systems are considered on the basis of (i) owner built vs. manufactured, (ii) batch vs. continuous flow, and (iii) large vs. small.

(i) Owner built vs. Manufactured

Owner built composting toilets are cheaper than manufactured units. While manufactured composters sell from upwards of $2,300, parts for a small owner built unit like the Pickle Barrel Batch can be purchased for as little as $200, less if the ventilation fan is dispensed with. A handy person with basic tools could construct such a unit in less than a day. Materials for the larger concrete block Minimus cost about $500 with a construction time of about 5 person days. The basic Minimus configuration has been successfully replicated with other materials, notably a fibro-lined stud frame, with reported savings in cost and time. Intermediate between the Pickle Barrel and the Minimus in both cost and degree of difficulty is the Farralones Batch system. Materials for the standard concrete block Farralones unit cost about $300 and the level of skill required is lower than for the Minimus. It was interesting to note that none of the owner builders surveyed resented the time spent constructing their unit. All enjoyed the experience and took pleasure in the fact that they had been able to contribute to the closing of nutrient cycles. On the other hand manufactured systems have the advantage of being easily installed and supported by the manufacturer's warranty. Most are relatively compact.

(ii) Batch vs Continuous Flow

Batch systems appear to require more maintenance operations than continuous flow units and need to be primed after each restart. Whereas the larger continuous flow units allow considerable flexibility in timing of the emptying operation, the generally smaller batch systems require immediate attention when the active chamber is full. Moisture holding capacity of the heap is also reduced during the start-up time in batch systems. Wheelie Batch owners, experiencing "fill-up" times of 3 to 4 months will possibly need to obtain additional chambers if they are to achieve the minimum 12 months residence time required by the NSW Guidelines. One Wheelie Batch owner found the change-over process difficult for one person to manage alone. By virtue of its larger size the Farralones Batch unit has longer fill-up times and hence longer intervals between change-overs.

(iii) Large vs. Small

Table 2 shows that all batch systems except for the Farralones are classified as small (chamber capacity <500L). These units have the advantage of compactness and can hence fit into smaller spaces. A criticism of the standard Minimus design is that its height of 2 metres makes it unsuitable for many flat sites. The much lower Rotaloo is probably the most suitable unit for a site where vertical clearance is restricted. The trade off in this case is the power needed to run the fan (20 watts) and the heater (250 watts). The Dowmus has overcome the problem of vertical clearance by going underground where necessary. This is made possible by using an auger to lift the final product out of the composting chamber. In general the larger systems were found to be more likely to perform well under passive ventilation. This is a considerable advantage given that owners of most small systems had experienced fan failure at some time.

Owners of large systems were found to be more likely to compost kitchen scraps and other organic waste in the toilet presumably because of fewer concerns in relation to fill-up time. It is thought that the bigger heap, particularly in the large continuous flow systems, is the major factor which makes these units more capable of handling peak loads without adverse effect.

Inputs and Outputs

It was found that owners used a variety of bulking materials including sawdust, wood shavings, grass clippings and paper either shredded or scrunched up. Some users attributed problems such as excess moisture and vinegar flies to the use of kitchen scraps. The solution was either to use less of this resource or to ensure that scraps were stored in a fly-proof container prior to addition to the toilet. In general owners of units with short residence time stored their final product in a closed container before using it. Only two owners applied final product to vegetable growing areas, most preferring to use it only under ornamentals or fruit trees.

Facilitating the Evolution of Composting Toilet Technology

Pollard (1997) makes a number of recommendations relating to specific modifications to the various composting toilet models examined in the study. Recommended design modifications include sculpting the floor of the Minimus to ensure flow of fluid to the drainage hole, and modification of the Farralones door to allow inspection of the heap without spillage of composting material.

Relevance of Study to Objectives of this Conference

Speaking about the interlocking issues of water supply and sanitation in developing countries Dr John Briscoe Senior Water Adviser to the World Bank makes the point that: "1,000 million still do not have ready access to safe water, and 2,000 million do not have sanitation. In fact, in urban areas, sanitation has actually declined", (Anon, 1997a). In a similar vein, Dr Richard Helmer, Chief of the World Health Organisation's Urban Environmental Unit predicts that roughly 60% of the world's population will live in mega-cities by the year 2025 (Anon., 1997b). Dr Helmer points out that 80%of Asians rely on groundwater and the greatest threat to the viability of the resource is microbiological contamination.

Two of the stated objectives of this conference are:

- "to develop a new range of innovative environmental technologies to meet the challenge of small scale decentralised treatment"; and further
- that "such units should also be able to be operated with minimum maintenance and supervision" .

The question is : "to what extent can composting toilet technology play a role in meeting these objectives and addressing the issues raised by Drs Helmer and Briscoe?"

It is probably the larger, passively ventilated owner built designs like the Minimus and Farralones Batch, with their greater tolerance for a variety of bulking materials and forgiveness in relation to promptness of emptying, which show most promise of adapting to the needs of urban and village dwellers in developing countries. However, in suggesting that this technology may be transferred to those situations, it must be stressed that the success of composting toilets in northern NSW owes a lot to (i) a cultural ambience which has encouraged "eco-friendly" technologies for over two decades, (ii) a supportive regulatory authority, and (iii) a well organised informal information network. Any attempt to establish composting toilets in places where they have not previously been used would have to be informed by an understanding of the need for similar support. Indeed this need to consider cultural issues was foreseen by Witold Rybczynski, (Director of the Minimum Cost Housing Group at the School of Architecture, McGill University, Montreal) who designed and built the first concrete block Minimuses in 1972. Although he had the sanitation needs of urban residents of developing countries in mind when he adapted the Clivus Multrum configuration, he was aware of the cultural dimension to the issue of technology transfer. His words of over 20 years ago are as relevant today as they were then.

"It should be stressed that there is no "one" design for the Minimus. It must be adapted to meet local climatic conditions, available building materials, local skills, and conditions. The application of composting sanitation technology to developing countries cannot be on a piecemeal basis. It must be done on a community (not individual) scale and integrated with social and educational development. . . . The use of the Minimus represents a significant shift in cultural patterns and habits and must be clearly understood by the users in order to be successful." (Hupping-Stoner, 1977)

Conclusion

In general the composting toilet systems studied appeared to be performing well. Owners showed a high degree of satisfaction with the technology. Any management problems that had arisen were overcome relatively easily by minor structural or behavioural modification. Owners also showed a generally high degree of interest in the technology and a willingness to experiment in order to optimise system performance. Each basic design appears to have its strengths and weaknesses. The smaller batch systems require less space but use more power and require more maintenance. The larger systems have the advantage of working well under passive ventilation and their larger heaps make them more able to cope with peak loadings and a wider variety of organic material inputs. These factors make them the most likely candidates for application in rural and urban areas of developing countries.

Although the success of composting toilets in rural areas of a developed nation may point the way to the transfer of the technology to developing countries, any attempt to do so should be informed by a thorough understanding of local cultural issues and be backed up by technical and educational support at a community level.

Acknowledgment

The authors wish to thank the thirty people who participated in the survey. Without your willing and enthusiastic assistance this study would not have been possible.

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RIPARIAN MARGINS AS BUFFER ZONES AGAINST NUTRIENT CONTAMINATION IN RECEIVING WATERS: NITROGEN, PHOSPHORUS AND SULPHUR BIOGEOCHEMISTRY IN RIPARIAN SOILS

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ABSTRACT

Riparian margins, acting as interface between uplands and aquatic ecosystems, are often used as buffer zones against nutrient contamination in receiving waters. In grazed pastures receiving regular fertiliser (superphosphate) applications and urinary nitrogen from grazing animals, significant amounts of phosphorus (P), nitrogen (N), carbon (C) and sulphur (S) may be lost from fertilisers, plant litter, animal excreta or soil particles via runoff or leaching. These may have a greater effect on the nutrient removal capacity of riparian wetlands receiving nutrient inputs from sheep grazed pastures than those receiving lower nutrient inputs from native bushes or mature pine forest. The objective of this study was to investigate the effects of these 3 types of land use on nitrification-denitrification potential, phosphate and sulphate retention capacity, sodium bicarbonate ($NaHCO_3$)-extractable P, hydriodic acid (HI)-reducible S, calcium phosphate ($Ca(H_2PO_4)_2$-extractable S, and organic C in riparian soils. Analyses were carried out on 9 soil cores taken from each riparian wetland to 3 different depths (0-50, 50-150 and 150-250 mm) during spring, summer, autumn and winter. Dissolved nutrients in waters passing through wetlands were also determined. At all sites, denitrification potentials were 50 times higher than nitrification potentials, suggesting that riparian soils are much less efficient at removing ammonium and organic N than nitrate. Different land use appeared to have no significant effect on soil microbial N cycling (i.e., soil nitrification-denitrification potentials) and soil phosphate retention capacity in riparian wetlands. At all sites, soil nitrification-denitrification potentials were highest in summer, while the reverse occurred for soil phosphate retention capacity, suggesting that the efficiency of riparian wetlands in stripping N or retaining P may vary with the seasons. Although the soil sulphate retention capacity in these riparian soils was low, HI-reducible soil S and $Ca(H_2PO_4)_2$-extractable soil S were higher in the pastoral catchment than those in native and pine catchments, indicating that riparian soils have a capacity to retain sulphate originating from superphosphate application and animal excreta in runoff. The absence of any consistent trend in dissolved nutrients, $NaHCO_3$-extractable soil P, and soil phosphate retention capacity across the riparian wetlands at all sites, indicated that other factors such as the interaction between hydrology and biogeochemistry may have an important influence on the ability of wetlands to act as buffers against nutrient contamination in receiving waters.

KEYWORDS

Riparian; biogeochemistry; nutrient removal; nitrification-denitrification; phosphate removal; sulphate retention; nonpoint source pollution

INTRODUCTION

Riparian margins, acting as interface between uplands and aquatic ecosystems, are well-recognised for their role in removing pollutants (sediment and nutrients such as nitrogen and phosphorus) from a variety of land use impacts (Osborne and Kovacic, 1993; Smith, 1989; Cooper *et al.,* 1995). Most studies assessing the efficiency of riparian zones in removing these pollutants have focused on the balance sheet of pollutant inputs and outputs from riparian wetlands (Osborne and Kovacic, 1993). This mass-balance approach is useful in providing information on the applicability of riparian wetlands in a particular situation. However, it does not provide adequate information on processes and factors that influence the effectiveness of riparian functions in pollutant removal, and the long-term sustainability of these processes (Cooper *et al.*, 1995). This study aims to investigate N and P cycling processes in three riparian areas that have been under different human intervention (native bush, exotic pine forest and intensively sheep-grazed ryegrass -white clover pastures with annual superphosphate applications). The study is part of a major research programme investigating land-riparian-stream interaction in New Zealand (Smith *et al.*, 1993). Although S is not considered a pollutant in creating eutrophication in receiving waters, investigation of soil S fractions at the three study sites would yield interesting information on the ability of riparian zones to intercept S from surface runoff and subsurface flows, since S is a major component of superphosphate fertiliser which is applied annually to the pasture riparian site. Numerous studies in New Zealand have indicated that S as sulphate or organic S may be lost from grazed pastures through leaching or runoff (Nguyen and Goh, 1994).

MATERIALS AND METHODS

Sites

The study sites (Table 1) are located within 15 miles from Hamilton, North Island, New Zealand.

Table 1. Some characteristics of the pasture, native and pine forest riparian sites

Characteristics	Pasture site	Native site	Exotic pine forest
Soil type	Waingaro steepland soil	Kaawa silt loam-Dunmore silt loam	Kaawa silt loam
New Zealand soil group	Yellow brown earth	Yellow brown earth-Yellow brown loam	Yellow brown earth
USDA taxonomy	Umbric Dystrochrept	Ochreptic Hapludult and Typic Hapludand	Ochreptic Hapludult
Vegetation	Ryegrass (*Lolium perenne*) and clover (*Trifolium repens*)	Ferns (*Leptospermum coparium, Dracophyllum subulatum and Blechnum penna-marina*) and native shrubs (Parataniwha, broadleaf, nikau tree ferns).	Pine (*Pinus radiata*) and some remnant pasture

The three catchments above the riparian sites are dominated by steep (>30°) to hilly (17-20°) topography (Smith *et al.*, 1993). Average annual rainfall (1600 mm) is reasonably distributed throughout the year, but tends to be lower in summer (January-March) than winter months (June-August). The native riparian zone is undeveloped and comprises of native shrubs and ferns. The pasture riparian zone is a part of pasture catchment which has been developed from the conversion of a native bush to sheep-grazed pasture since the 1960s. Superphosphate with an average phosphorus (P) and sulphur (S) content of approximately 9.3 and 11% was applied in summer (December-January) each year for at least 28 years at the rate of approximately 300 kg /ha/year, providing annual sulphur (S) and phosphorus (P) inputs of 29 kg P/ha and and 33 kg S/ha respectively. Pasture hebage is predominantly ryegrass (*Lolium perenne*)-white clover (*Trifolium repens*) grazed by sheep throughout the year, with an average stocking rate of 13 stock units (SU)/ha. One SU is equivalent to a ewe (55 kg liveweight at mating) plus one weaned lamb per year. Each SU consumes 550 kg dry matter annually (Nguyen and Goh, 1994).

With an annual stocking rate of 13 SU/ha and a significant proportion (20%) of N-fixing clover in mixed pasture herbage, N leaching losses from urine patches is expected to occur at times (e.g., winter months) when rainfall exceeds water-holding capacity. Soil parent materials in all three catchments are predominantly sedimentary sandstones and siltstones (greywacke and argillite).

Field-moist soils (9 soil cores; 25 mm diameter) from each riparian site were taken in late winter/early spring (August-November) and summer (February) to a depth of 250 mm and subsequently divided into 3 sections (0-50; 50-100 and 100-250 mm). These samples were wet sieved (< 2 mm) and stored at 4^{o} C until analysis (usually < 4 days after storage) for labile soil P, adsorbed plus soluble S, soil sulphate and phosphate retention capacities. Subsamples were air-dried, and finely ground (<0.150 mm) before analyses for soil total C, total N, and hydriodic acid-reducible S.

Analyses

Soil moisture content was determined gravimetricaly by drying after oven-drying subsamples for 48 hours at 105^{o} C. Bulk density of soil samples was determined as the ratio of oven-dried weight (g) of soil over the volume (cm^3) of soil collected by the soil sampler.

Field-moist soil samples were extracted for adsorbed plus readily-soluble S with 0.01 M $Ca(H_2PO_4)_2$ (1:5 soil:solution ratio for 1 hour; Barrow, 1967), labile soil P with 0.5 M $NaHCO_3$-extractable P (1:5 ratio for 30 minutes; Olsen and Sommers, 1982) and KCl-extractable ammonium and nitrate (1:10 soil:solution ratio for 1 hour; Keeny and Nelson, 1982).

Soil phosphate retention capacity was measured by shaking 5 g samples of field moist soil for 16 hours with 25 mL of 0.2 M $NaHCO_3$ solution at pH 4.65, containing 1000 mg P/L (Saunders, 1965). The amounts of P left in the soil extracts were determined by the Murphy and Riley method (1962). The soil phosphate retention capacity was then calculated as the percentage of added P that was sorbed by soils. If soil phosphate retention capacity exceeded 80-90%, pore water was removed from another set of soil samples (5 g on oven-dried basis) by centrifugation (20 minutes at 5,000 rpm) and analysed for dissolved phosphate (DP) using the Murphy and Riley method (1962). After pore water was removed, soil samples were shaken (120 rpm) with 25 mL of distilled water for 24 hours. Water-soluble phosphate was then determined using the Murphy and Riley method (1962).

Soil sulphate retention capacity was measured by shaking duplicate 5 g samples of moist soils for 16 hours, one sample with 25 mL 0.01 M $CaCl_2$ solution and the other with 25 mL 0.01 M $CaCl_2$ containing 50 mg S/L (Saunders and Hogg, 1971). The amount of S remaining in the $CaCl_2$ soil extracts was assayed colorimetrically using the Johnson-Nishita procedure (1952). The soil sulphate retention capacity was then calculated as the percentage of added S adsorbed by the soils after taking into account the amount of native soil S extracted by the $CaCl_2$ reagent.

Hydriodic acid reducible soil S (HI-reducible S) fraction was determined by the method of Freney *et al.* (1969). and ester sulphate was estimated as the difference between HI-reducible S and $Ca(H_2PO_4)_2$-extractable soil S (Freney *et al.*, 1969).

Total C and total N were determined by the combustion method (Perkin-Elmer 2400 CHN Elemental Analyzer), while nitrifying and denitrifying potentials were determined using the methods described in Cooper (1986).

RESULTS

Soil carbon and nitrogen

Soil C and N content was higher in pastoral riparian site than that in native riparian or exotic pine forest, while soil C:N ratios were lowest in pastoral riparian soils (Table 2). At all sites, soil C and N decreased with soil depth and C:N ratios at all depths were less than 21:1 (Table 2). There was no consistent trend in the change in soil C, N and C:N ratios with seasons. Soil bulk density increased with soil depths but it did not change with seasons (Table 2).

Table 2. Soil carbon and nitrogen contents and soil bulk density (means ± standard errors; *n* = 9) at three different depths in three study riparian sites

Site	Soil depth (mm)	C (%)	N (%)	C : N	Bulk density (g/cm^3)
		Early Spring (August)			
Pasture	0-50	9.1 (0.20)	0.90 (0.024)	10.1 (0.09)	1.15 (0.027)
	50-100	6.9 (0.09)	0.64 (0.023)	10.8 (0.27)	1.22 (0.012)
	100-250	3.7 (0.17)	0.35 (0.023)	10.7 (0.25)	1.37 (0.027)
Native	0-50	8.4 (0.36)	0.44 (0.075)	20.8 (5.27)	1.18(0.019)
	50-100	6.6 (0.28)	0.39 (0.082)	14.1 (1.23)	1.25 (0.018)
	100-250	2.8 (0.39)	0.38 (0.108)	14.7 (1.19)	1.47 (0.024)
Pine	0-50	6.0 (0.61)	0.49 (0.035)	12.2 (0.49)	1.24 (0.036)
	50-100	4.3 (0.51)	0.39 (0.038)	11.2 (0.24)	1.38 (0.027)
	100-250	2.6 (0.05)	0.24 (0.009)	10.5 (0.18)	1.51 (0.026)
		Summer (February)			
Pasture	0-50	8.2 (0.12)	0.82 (0.009)	10.1 (0.21)	1.10 (0.038)
	50-100	7.0 (0.61)	0.66 (0.057)	10.7 (0.23)	1.16 (0.032)
	100-250	4.8 (0.79)	0.44 (0.069)	10.8 (0.13)	1.27 (0.049)
Native	0-50	7.8 (0.13)	0.51 (0.009)	15.2 (0.46)	1.17 (0.015)
	50-100	6.1 (0.48)	0.42 (0.029)	14.4 (0.41)	1.24 (0.019)
	100-250	3.1 (0.43)	0.26 (0.032)	12.1 (0.26)	1.40 (0.023)
Pine	0-50	5.7 (0.58)	0.49 (0.044)	11.6 (0.23)	1.22 (0.033)
	50-100	4.2 (0.33)	0.33 (0.020)	12.7 (1.34)	1.34 (0.012)
	100-250	2.6 (0.14)	0.25 (0.014)	10.7 (0.08)	1.50 (0.012)

Table 3 shows that soil ammonium (KCl-extractable ammonium) in pastoral riparian site was higher than that in native riparian or exotic pine forest. At all sites, soil ammonium was a dominant form (>96%) of KCl-extractable inorganic N regardless of seasons and soil depths. Soil ammonium was higher in early spring than summer and decreased sharply with depth. Compared to ammonium, soil nitrate at all sites was always less than 2 mg/kg soil and did not consistently decrease with depth.

Table 3. Potassium chloride-extractable ammonium and nitrate concentration and proportion of extractable inorganic nitrogen as ammonium (means ± standard errors; n = 9) in riparian soils at three different depths in spring and summer months

Riparian site	Soil depth (mm)	Ammonium (mg N/kg soil)	Nitrate (mg N/kg soil)	Ammonium (% KCl-extractable inorganic N)
		Early Spring (August)		
Pasture	0-50	507 (99.8)	0.94 (0.037)	99.8 (0.03)
	50-100	247 (3.6)	0.89 (0.048)	99.6 (0.02)
	100-250	99 (6.8)	0.93 (0.111)	99.0 (0.15)
Native	0-50	162 (6.5)	0.71 (0.121)	99.6 (0.09)
	50-100	115 (32.7)	1.02 (0.152)	99.1 (0.13)
	100-250	101 (24.7)	1.31 (0.263)	98.5 (0.51)
Pine	0-50	200 (14.8)	0.94 (0.091)	99.5 (0.08)
	50-100	122 (8.9)	0.89 (0.049)	99.3 (0.03)
	100-250	85 (6.9)	1.05 (0.043)	98.7 (0.13)
		Summer (February)		
Pasture	0-50	318 (30.3)	0.76 (0.077)	99.7 (0.04)
	50-100	171 (12.6)	0.57 (0.089)	99.7 (0.07)
	100-250	122 (7.5)	0.48 (0.173)	99.6 (0.12)
Native	0-50	86 (12.0)	0.85 (0.183)	99.0 (0.12)
	50-100	71 (10.0)	1.49 (0.118)	97.9 (0.27)
	100-250	73 (3.8)	1.20 (0.370)	98.4 (0.53)
Pine	0-50	82 (4.9)	0.85 (0.187)	98.9 (0.31)
	50-100	44 (6.6)	1.07 (0.214)	97.5 (0.31)
	100-250	42 (2.5)	1.47 (0.148)	96.4 (0.59)

Soil nitrification and denitrification potentials are presented in Figures 1 and 2, respectively. At all sites, soil denitrification potentials in the top 50 mm depth increased by a factor of 5 from late winter/early spring to summer (Figure 2) and soil denitrification potentials (Figure 2) were 50 times higher than nitrification potentials (Figure 1). In each sampling season, there appeared to be no difference in microbial N activity (nitrification-denitrification) between the 3 study riparian sites over the entire 250 mm soil depth.

Soil phosphorus

Table 4 shows that labile soil P was higher in riparian pasture than native or riparian forest sites in late winter/early spring. However, this difference was not pronounced in summer. Soil phosphate retention capacity was above 70% for all sites during late winter/early spring. However, soil phosphate retention capacity was extremely low for all sites in summer. Soil moisture content in late winter/early spring and summer was approximately 43-84% and 25-43% respectively (Table 4).

Pore water collected from all sites during late winter/early spring sampling period (when soil phosphate retention of > 80%) had low phosphate concentration of 0.02-0.46 mg P/L (i.e., ranging from 0.1 to 1.9 mg P/kg soil). Water-extractable P from soil samples collected during this period was also low, ranging from 0.7 to 2.3 mg P/kg soil (Table 4).

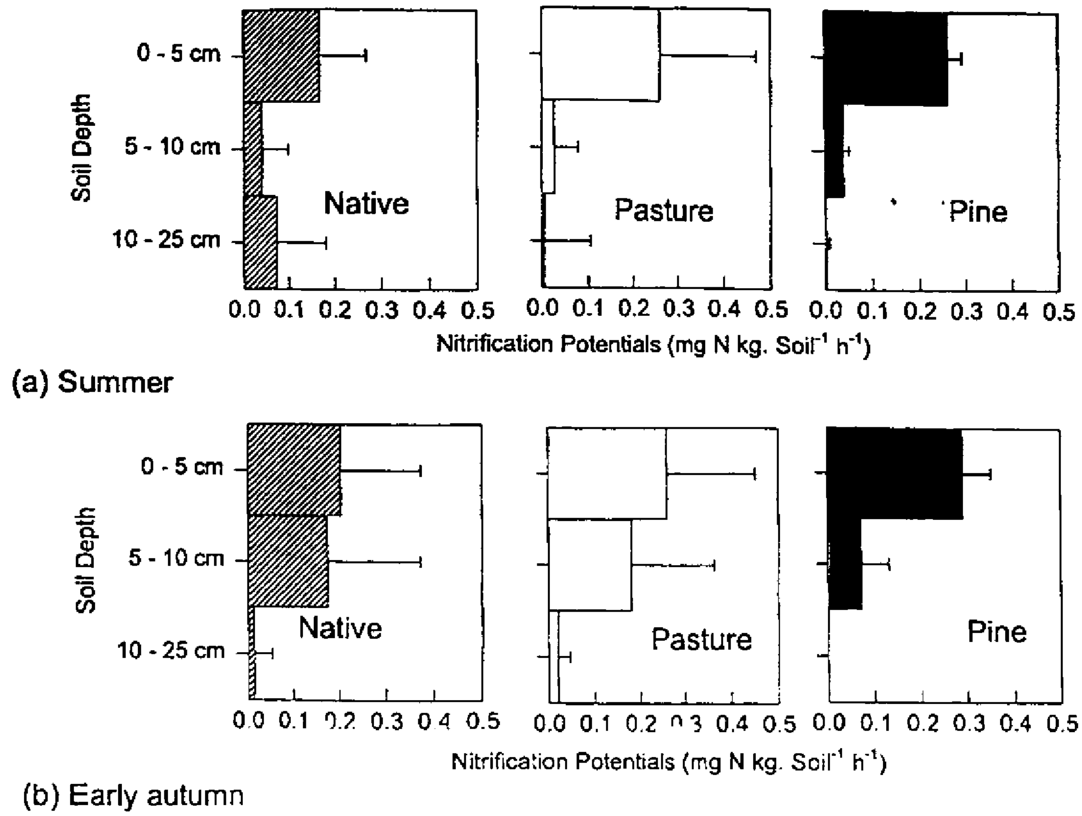

Figure 1. Nitrification potentials in riparian soils in native , pasture and pine catchments in summer (a) and early autumn (b). Error bars = standard errors of the mean (n=6)

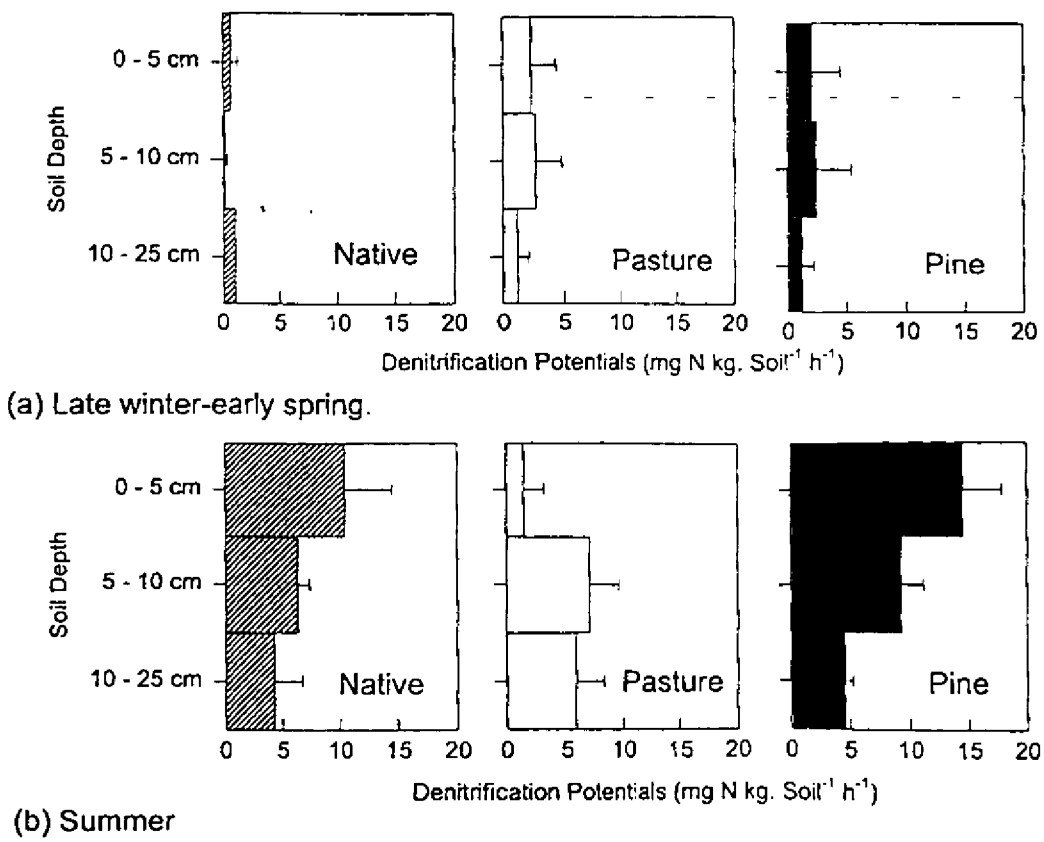

Figure 2. Denitrification potentials in riparian soils in native , pasture and pine catchments in late winter-early spring (a) and summer (b). Error bars = standard errors of the mean (n=6)

Table 4. Phosphate retention capacity, labile soil phosphorus (P), pore water P, water-extractable P, and moisture content in soils (means ± standard errors; n = 9) at three different depths for three riparian sites in late winter/early spring and summer months

Riparian site	Soil depth (mm)	P retention (%)	Labile soil P (mg P/kg soil)	Pore water P (mg P/kg soil)	Water-extractable P (mg P/kg soil)	Moisture (%)
			Early spring			
Pasture	0-50	94 (1.7)	120 (5.1)[a]	1.1 (0.39)	0.7 (0.20)	84 (1.3)
	50-100	80 (6.3)	25 (3.6)	0.4 (0.14)	0.9 (0.17)	71 (1.0)
	100-250	86 (4.3)	13 (0.6)	0.3 (0.05)	1.7 (0.49)	56 (2.9)
Native	0-50	72 (6.4)	20 (1.5)	0.6 (0.15)	1. 5 (0.05)	72 (0.6)
	50-100	87 (5.5)	10 (0.7)	0.5 (0.10)	1.8 (0.11)	67 (1.8)
	100-250	94 (0.6)	10 (0.5)	0.3 (0.02)	2.3 (0.78)	48 (1.5)
Pine	0-50	75 (7.8)	15 (1.2)	0.6 (0.06)	1.2 (0.11)	71 (1.1)
	50-100	76 (2.0)	11 (1.0)	0.5 (0.10)	1.6 (0.09)	54 (2.3)
	100-250	81 (0.3)	7 (0.5)	0.3 (0.02)	2.3 (0.19)	43 (1.3)
			Summer			
Pasture	0-50	13 (1.3)	20 (0.7)	ND	ND	43 (1.2)
	50-100	20 (0.8)	14 (0.8)	ND	ND	35 (1.7)
	100-250	31 (1.4)	11 (0.8)	ND	ND	30 (1.8)
Native	0-50	25 (1.1)	17 (0.7)	ND	ND	40 (1.3)
	50-100	24 (1.2)	15 (0.6)	ND	ND	37 (0.7)
	100-250	34 (3.3)	12 (0.4)	ND	ND	32 (1.0)
Pine	0-50	15 (1.4)	12 (0.2)	ND	ND	37 (0.9)
	50-100	8 (0.4)	9 (0.4)	ND	ND	29 (1.4)
	100-250	9 (0.2)	7 (0.4)	ND	ND	25 (1.0)

ND, not determined.

Soil sulphur

Unpublished data (Nguyen, 1997) shows that soil sulphate retention capacity at all depths for all sites was low (< 32%) during late winter/early spring and the summer months. In late winter/early spring samples, pastoral site had up to 1300 mg S/kg soil as ester sulphate while native and riparian sites had less than 400 mg S/kg as ester sulphate. Similarly, ester sulphate in pastoral sites was higher than that in native or pine sites even in the subsoil (100-250mm) layer (565 ± 186, 258 ± 9, and 114 ± 6 mg S/kg respectively). There was no consistent trend in (CaH_2PO_4)-extractable S with depth while ester sulphate at all sites decreased with depth. $Ca(H_2PO_4)_2$-extractable S accounted for 96% of ester sulphate in late winter/early spring but less than 30% in summer.

DISCUSSION

The high soil C content at all sites suggests that a supply of readily-soluble labile C (energy source) from the studied riparian soils is probably sufficient for the denitrification process (Hill, 1996; Willems *et al.*, 1997). The low soil nitrate content in the three study sites may be the result of denitrification (Schipper *et al.*, 1993) and dissimilative nitrate reduction to ammonium (Schipper *et al.*, 1994). The high denitrification potentials in the three study sites (50 times higher than nitrification potentials; Figures 1 and 2) may account for the higher accumulation of ammonium compared with nitrate. Alternatively, the high ammonium in these soils may indicate that mineralisation of organic N is incomplete and that ammonium is not readily nitrified to nitrate under anaerobic conditions. Several studies (e.g., Enzewor, 1976; Bowden, 1984) have reported that ammonium is produced as a result of anaerobic decomposition and mineralisation in highly organic soils with C:N ratios of less than 30:1. Although denitrification potentials are high in these soils after nitrate amendment to ensure favourable conditions for denitrification, the actual denitrification rate may be potentially limited by the low nitrate at all sites (Hill, 1996). Nitrification of ammonium (derived from surface runoff or organic N mineralisation) is therefore likely to be a major factor controlling denitrification at the study sites. Since nitrate is an end product of nitrification and a primary substrate for denitrification process (Hill, 1996), the low soil nitrate content may not mean that limited nitrification or denitrification occurred in these riparian soils. It may be due to the rapid nitrate denitrification to nitrogen gases as soon as nitrate is produced by nitrification. Nitrate leaching beyond a 250 mm soil depth was also another possibility, since nitrate is a weakly adsorbed anion and readily mobile in water (Haynes, 1986). In-situ denitrification and nitrification need to be carried out in these soils to determine the relative importance of these processes and nitrate leaching losses in N dynamics in riparian wetlands and to assess the effect of low soil nitrate content on denitrification.

At all sites, soil denitrification potentials increased by a factor of 5 from late winter/early spring to late summer, suggesting that nitrate removal from surface runoff and subsurface flow is lower during winter months, when the soil temperature may be unfavourable for soil microbial activity.

The high soil C and N content at all riparian sites, particularly in pastoral soils, is attributed to a regular return of C and N from plant residues. The higher soil C and N accumulation in pastoral riparian sites could have resulted from an adequate supply of P and S from superphosphate applications enhancing ryegrass-clover production, symbiotic nitrogen fixation and possibly microbial immobilisation of soil organic C and N (Sarathchandra *et al.*, 1984; Nguyen and Goh, 1992).

The critical soil C:N ratio below which net N mineralisation occurs is 20:1 (Haynes, 1986). Since soil C:N ratios (Table 2) were normally below 20:1, N mineralisation was expected to occur at all sites if aerobic soil conditions existed. Management practices could be implemented to enhance N removal by modifying hydrologic conditions so that soil ammonium is readily nitrified to nitrate for subsequent denitrification.

The higher labile soil P content in riparian pasture, compared with that in native and riparian forest in late winter/early spring, may reflect the capacity of riparian soils at the pasture site to retain dissolved P (DP) from surface runoff during the winter months, when soil moisture content may be sufficient to create anaerobic conditions. Losses of DP in surface runoff probably originated from superphosphate fertilisers, animal excreta and the release of a water-soluble P fraction from plant residue. Phosphate retained in riparian soils may subsequently be soluble in runoff water and a source of DP to streams and rivers during summer runoff events, since soil P retention capacity in all study sites was less than 40% in summer. Riparian soils may therefore be either a sink or source of DP to receiving waters, depending on soil phosphate retention capacity. Cooper *et al.* (1995) raised a similar issue when they reported that riparian set-aside (12 years of retirement) in a New Zealand farming system led to the development of a riparian zone that was enriched in labile soil P and could be a source of DP to receiving waters, compared with that in riparian grazed pastoral site and riparian native soil.

The increase in soil P retention during late winter/early spring may result from a decrease in soil redox (reduction-oxidation) potential and an increase in the water:soil ratio (Reddy *et al.*, 1995; Nguyen *et al.*, 1997). Phosphate retention has been found to be higher under anaerobic than under aerobic conditions (Patrick and Khalid, 1974; Nguyen *et al.*, 1997), probably due to the reduction of ferric oxyhydroxide to more soluble ferrous forms under reducing conditions, thus increasing phosphate sorption sites and overall P sorption (Reddy *et al.*, 1995). Evidence of strong phosphate retention capacity of riparian soils in late winter/early spring was further shown by the low level of DP in soil pore water (0.02-0.46 mg/L) collected during this period. Furthermore, these soils released less than 2.3 mg/kg soil after 24-hour extraction with distilled water, indicating that phosphate is tightly held by soil particles.

The higher accumulation of organic S forms in the pastoral riparian site, compared with that in native and forest sites probably reflects a high contribution of sulphate and organic S from the pastoral catchment receiving annual superphosphate application and the regular return of animal excretal S and plant residues. Sulphate and organic S from either surface runoff or subsurface flow may be trapped in pastoral riparian soils by incorporating into the soil organic S pool. Microbial S immobilisation is likely to be the major feature of S retention in riparian soils, since these soils have a very low capacity to adsorb sulphate against leaching (sulphate retention capacity ranging from 2-32%) or lateral runoff. The higher amount of ester sulphate found in the top 250 mm of soil in the pastoral riparian site suggests that organic S in ester sulphate form has been leached down the soil profile. Since some labile soil organic S (ester sulphate) may be included in the $Ca(H_2PO_4)_2$-extractable S fraction (Nguyen and Goh, 1992), the increase in the $Ca(H_2PO_4)_2$-extractable S fraction in late winter/early spring may reflect the increase in labile soil organic S fraction during the period.

CONCLUSION

Land use does not seem to affect the N and P removal capacity of riparian wetlands in pastoral, native and pine forest catchments. Regardless of land management practices, N removal capacity is expected to be higher in summer than winter, while the reverse occurs for P removal. If ammonium in riparian soils can be readily nitrified to nitrate during summer months, not only nitrate from subsurface flow but ammonium and organic N from surface runoff is potentially denitrified in riparian soils at these sites. Riparian soils can be a source of P to receiving waters in summer if there is a runoff event during this period because of the low P retention capacity in these soils. These seasonal effects on N and P removal in riparian wetlands are likely to be attributed to the interaction between hydrology and N and P cycling in riparian soils. Further research is required to investigate this interaction. Soil microbial activity may play an important role in the cycling of C, N, P and S and hence the nutrient removal function of riparian wetlands.

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SUITABILITY OF THE H_2S PAPER STRIP METHOD FOR TESTING RAINWATER

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ABSTRACT

Rainwater is the source of drinking water for many people in Australia and in other parts of the world. The water that is collected in the tanks may contain a variety of bacteria of bird and reptile faecal origin. The rainwater tanks are seldom treated and these bacteria can be of human health concern. The quality of rainwater is rarely tested due to multiple reasons. The availability of a cheap testing procedure, which could be conducted by the householders, would be advantageous. The H_2S paper strip method is an on - site bacterial testing method which detects the hydrogen sulphide producing bacteria in the water sample. A good correlation has been reported between the presence of hydrogen sulphide producing bacteria and the faecal indicators. In order to assess the suitability of the H_2S method for testing rainwater, 113 samples were analysed for total coliforms, *Escherichia coli* and *Salmonella* sp by the standard procedures and the results compared with the H_2S method. Coliforms were observed in 59 samples and 32 samples contained *E.coli*. The H_2S method gave more true results when the bottles were incubated at 24 hours than at 48 hours. Many false positive and false negative results were observed. The false positive results could be due to the presence of other Enterobacteriacea such as the *E.cloacae*, *Proteus*, *Citrobacter* and *Salmonella arizona* as well as the H_2S producing bacteria of plant origin which were isolated from some of the positive bottles. The presence of false negative results is a concern in using this method as an authentic test for testing drinking water but it could be used as a screening test for isolated drinking water sources.

KEYWORDS

Rainwater, H_2S method, coliforms, *E.coli* and *Salmonella sp.*

INTRODUCTION

Rainwater is a source of drinking water for many householders throughout Australia as well as other parts of the world. Rainwater is considered to be a pure source of drinking water and therefore good to drink even without treatment. However there are many sources of contamination which may affect the physical, chemical and microbial quality of the water stored in tanks. Reptile and bird droppings as well as the decayed plant and animal matter carried to the tank with the rainwater are the main sources of contamination. The presence of overhanging trees increases the risk of bird droppings. Therefore even if the tanks are properly covered the rainwater in the tank may get contaminated. This is often left unnoticed and rainwater tanks are seldom treated. The microbial quality of rainwater stored in tanks was found to be very poor by Yaziz *et al.*(1989), Thomas and Greene, (1993), Fujioka *et al.*(1995), Rijal and Fujioka (1995). As each tank has to be considered as an individual unit, water quality assessment and treatment are primarily the interest of the householders. The water quality in the tank changes according to the season. Therefore there is a need for frequent testing of water quality in the tanks. However frequent monitoring of the quality of rainwater tank is difficult, as the standard methods currently available are expensive, that demand laboratory and technical support. The on - site testing methods such as the Colilert and the Colisure are expensive for routine testing especially by the householders. An affordable and easy method to test water by the householders is a requisite for testing rainwater tanks as well as for isolated drinking water sources.

The H_2S method developed by Manja *et al.* (1982) is an onsite bacterial testing method, which is simple and less expensive. This method detects the hydrogen sulphide producing bacteria rather than the faecal indicators or specific pathogens. The H_2S method was found to give a good correlation with the standard methods for detecting faecal pollution (Rijal *et al.*, 1995; Ziel *et al.*, 1995; Grant and Ziel, 1996; Martins *et al.*, 1996). Since a majority of *Salmonella* sp produce H_2S, the method was found to be good for testing the presence of *Salmonella* (Gawthorne *et al*, 1996; Pillai *et al.*, 1997). Many members of the Enterobacteriacea such as the *Citrobacter, Klebsiella and Proteus* produce hydrogen sulphide so their presence will also be detected by this method.

The present study was aimed at assessing the suitability of the H_2S method for testing rainwater. Results of the H_2S test were compared with the standard procedures for testing total coliforms, *Escherichia coli* and *Salmonella* spp. As rainwater from each tank is different, the samples taken from tanks formed a wide range to be analysed for testing the efficiency of the method. It was previously reported that addition of l-cystine to the H_2S medium improved the efficiency of the method (Venkobachar *et al.*, 1994; Pillai *et al.*, 1997). The study also compared the two media for testing rainwater.

MATERIALS AND METHODS

Rainwater samples (500ml) were collected from household tanks in and around Perth, Western Australia. Sterile bottles were used for collecting the samples. Samples were collected from the taps on the rainwater tank after running out the water for 10 seconds to prevent contamination from the taps. The samples were analysed within 24 hours of collection and were refrigerated if stored for more than 4 hours. The water was tested for total coliforms, *Escherichia coli* and *Salmonella* by the membrane filtration method described by the *HMSO* (1982). Each sample was simultaneously tested with the H_2S Bottles. The H_2S medium (M1) was prepared according to Manja *et al* (1982). A modification of the H_2S medium (M2) with 0.25g of L-cystine added to 100 ml of the medium was also tried. The bottles were incubated at 37°C. The bottles were examined after 24 and 48 hours for positive results which were indicated by the blackening of the bottle. The bottles that did not turn black after 48 hours were considered as negative. The data were analysed to find a correlation of the H_2S method with the presence of total coliforms, *E.coli* or *Salmonella* spp. Some of the bacteria that caused positive results were identified using 20E API strips.

According to WHO Guidelines (1996) treated drinking water should not contain any coliforms but the presence of coliforms in 5% of the total samples collected in a year is allowed. Australian drinking water guidelines NHMRC & ARMC (1996) recommend that 100ml of the sample should not contain any coliforms or *E.coli*.

RESULTS

The modified H_2S medium (M2) was tested with 113 rainwater samples and the original H_2S medium was tested for 53 of these samples. True positive results were taken as those that were positive for the H_2S test and positive for total coliforms or *E.coli*. True negative results are those which are negative for H_2S and total coliforms or *E.coli*. False positive are positive for H_2S and negative for total coliforms or *E.coli* while negative H_2S result for a positive total coliforms or *E.coli* are considered as false negative. Table 1 shows the percentage of true and false results with medium M1 and M2 at 24 and 48 hours of incubation. Out of the 113 samples analysed 59 samples contained coliforms and 32 samples contained *E.coli*. The H_2S method gave more true results when the bottles were incubated for 24 hours. After 24 hours of incubation 48 samples were positive and after 48 hours, 67 bottles turned positive. High contamination with total coliforms (>10CFU/100ml) was noticed in 35 samples. Many false positive and false negative results were observed with the two media. The reason for the false positive and false negative results could not be identified but *Enterobacter cloacae, Proteus and Citrobacter* were isolated from some of the positive bottles. *Salmonella arizona* was present in two samples where the tank was never cleaned or treated. Four samples which were false negative contained *E.coli*. Table 2 shows the total coliform and *E.coli* counts of the samples which were false negative.

Table 1. Percentage of true and false results with M1 and M2 at 24 and 48 hours

	No. of Samples	True Positive (H_2S &TC or *E.coli*+ve)	True Negative (H_2S &TC or *E.coli*-ve)	False Positive (H_2S+ve &TC or *E.coli*-ve)	False Negative (H_2S-ve &TC or *E.coli*+ve)
M1(24 hrs)	53	19 (35.85%)	22 (41.51%)	2 (3.77%)	10(18.87%)
M1 (48 hrs)	53	24 (45.28%)	17 (32.07%)	7 (13.21%)	5 (9.43%)
M2 (24 hrs)	113	42 (37.17%)	48 (42.48%)	6 (5.3%)	17 (15.04%)
M2 (48 hrs)	113	49 (43.36%)	36 (31.86%)	18 (15.93%)	10 (8.85%)

Table 2. Data of total coliforms and *E.coli* in false negative samples

No	Total coliforms(CFU/100ml)	*E.coli* (CFU/100ml)	M2	M1
1	38	0	neg	
2	4	3	neg	
3	25	8	neg	
4	2	1	neg	neg
5	1	0	neg	neg
6	1	0	neg	neg
7	31	16	neg	neg
8	5	0	neg	neg
9	1	0	neg	neg
10	14	0	neg	neg

With the medium M2 the percentage of false results increased when the bottles were incubated for 48 hours. At a total coliform count >0 CFU/100ml M2 gave more true results than M1 after 24 hours of incubation. But when incubated for 48 hours M1 gave slightly more true results than M1. The percentage of true results and false results with M1 and M2 at 24 and 48 hours are shown in

Fig. 1. True results are the total of true positive and true negative results whereas false results are the total of false positive and false negative results.

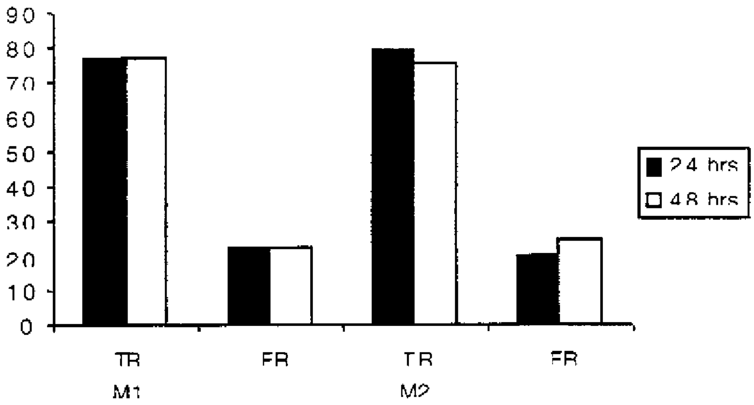

Fig.1. Percentage of true and false results for different media at 24 and 48 hours of incubation

DISCUSSION

Analysis of the water samples showed that about half of the rainwater tanks tested were contaminated with micro organisms of health concern. In medium M2, L-cystine accelerated H_2S production and therefore more positive results were observed at 24 hours whereas in M1 the H_2S production was slightly slower and therefore took 48 hours. Pillai *et al* (1999) observed that M2 was better than M1 for detecting faecal coliforms to the lowest concentration and at a wider range of incubation temperature. However the greater sensitivity of M2 is associated with a greater number of false positive results when compared to the total coliform and *E.coli* tests.

As the method detected H_2S producing bacteria, the false positive results could be due to the presence of other H_2S producing bacteria like *Enterobacter* sp., *Proteus* and *Citrobacter* which were isolated from some of the positive bottles. This was observed by Castillo *et al.* (1994). Although all samples were tested for *Salmonella* sp. only 2 contained *S. arizona*. Some phyto pathogens such as *Erwinia* sp and *Chromobacterium* were also isolated from positive bottles. The source of these bacteria could be leaves that were carried into the tanks. Many phytopathogens as well as the other bacteria identified from the positive samples are of human health concern. Therefore false positive may indicate greater risk of infection and the H_s S method may be a better indication of human health risk potential.

False negative results observed are a concern when using this method for testing rainwater. The results proved that false negative result is not dependent on the coliform count as even the sample with very high numbers of coliforms gave negative results. This showed that the method did not fully correlate to the presence of coliforms and some samples although contained coliforms did not contain the H_2S producing bacteria. Levett (1990) reported that in faeces H_2S producing bacteria occurred at a concentration of 10^{10}/g. In that case it seems likely that H_2S producing bacteria would be present in the samples contaminated with faeces. However the presence of *E.coli* in 4 samples that were false negative suggests that this may not always be the case. As to whether the negative results indicate a less human health risk potential is a question.

Ziel *et a.l* (1995) reported that the H_2S method had a 87.7 % correlation with the presence/absence (P/A) coliform test. Martins and Pellizari (1990) compared the H_2S Method with different coliform tests and reported a percentage agreement varying from 66.7-90% with raw waters and a 90-94 % with drinking water samples. More positive results were obtained with the H_2S test than the other tests as was found in the present work. According to the data given by Grant and Ziel (1996) out of 14 well water samples tested there was 1 false positive and 1 false negative result whereas 7.2% false negative results were observed by Castillo *et al.* (1994). The false positive and false negative results were also observed by Desmarcheliar *et al.* (1992).

Rijal and Fujioka (1995) tested 5 different sources of rainwater tanks for five weeks and observed that the concentration of total coliforms correlated well with H_2S producing organisms. They concluded that testing water for hydrogen sulphide bacteria is a reliable method for assessing the quality of water. But since the same five cisterns was tested for five weeks the bacterial population could almost be the same. Also the number of false positive and false negative results were not identified separately. As shown in Fig 1. with M2 79.64 % true results were obtained where as with M1 it was 77.35%. Therefore the addition of l-cystine to the medium would increase the correlation. M2 had 5.3% false positive and 15.03% false negative results while M1 had 3.77% false positive and 18.87% false negative results. Wallis (1991) reported that on testing rainwater tanks in Thailand 20 % false positive and 41 % false negative results were obtained. Hazbun and Parker(1983) recommended that false negative results could be reduced by increasing the incubation temperature to 37 °C for 24 hours and the numbers as done in the present work.

CONCLUSIONS

1. This method could be an initial screening test which could be carried out by the householders to asses the quality of rainwater.
2. The addition of l-cystine to the H2S medium would increase the correlation with the standard methods for testing coliforms.
3. The false positive results obtained could be due to Enterobacteriacea other than the coliforms or due to the presence of other H_2S producing bacteria of animal or plant origin.
4. The appearance of false negative results indicates that H2S producers may not always be present need to be investigated.

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Treatment and Modelling

EMPIRICAL WATER QUALITY MODELLING FOR UNSEWERED URBAN PLANNING USING PHYSIOGRAPHIC PARAMETERS

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ABSTRACT

Urban physiography, that is the type and extent of urban surfaces including for example impervious surfaces and related urban runoff coefficients, have long been associated with urban water quality and used for a variety of urban modelling purposes (eg. SWMM, HSPF). These parameters have also been attributed to affecting urban water quality, including surface and ground water quality in unsewered urban areas, and have been suggested by various authorities to be taken into account in the construction of environmental planning instruments such as total catchment management plans. Unfortunately, there are very limited data available, notably in Australia, to provide for the construction of reliable predictive relationships in unsewered urban areas.

A series of urban runoff monitoring investigations are described where pollutant concentration and load relations are developed for small urban areas on shale in the southwest of Sydney, NSW. These sites are used to generate generalised empirical runoff models which may be used to estimate both pollutant concentrations and loads in surface waters where relatively impermeable soils are encountered and surface water quality is potentially at risk. Rainfall related runoff factors are incorporated so that the models can be utilised for the development of urban planning controls, such as minimum allotment sizes, and housing density (ie. house.hectare^{-1}). A range of parameters are considered including suspended solids, bacteria, nutrients and heavy metals.

KEYWORDS

On-site Wastewater Management, Water Quality, Modelling

INTRODUCTION

Unsewered urban areas are where household wastewater is typically treated and disposed of on-site, usually somewhere within the owners property. This is in conflict with current urban planning thinking which favours populated regions to have access to main-line sewer, with reticulation to a remote treatment facility for health and environmental reasons. However, many urban areas, particularly outer city suburbs and small towns, still utilise on-site treatment technologies because of infrastructure costs.

Urban areas are physically diverse and occur as a continuum of types from residential housing to highly industrialised centres. Urban physiography, in the context of this work, refers to the physical attributes of the catchment including catchment area, housing density, impervious cover, road cover, curbage, grade and runoff coefficients. Sources of urban physiographic variability include (Ebise, 1990):

- development density;
- character of natural environmental setting of the urban unit; and
- the management of waste products imported into or generated within the drainage basin.

Numerous stormwater runoff models have been developed which attempt to simulate runoff and water quality from urban areas (eg. SWMM or Storm Water Management Model, Huber and Dickinson, 1988). However, none of these models have to date accommodated for the water and contaminant influxes from unsewered areas. This makes it difficult to utilise such process-based models for the development of urban planning controls in unsewered urban areas.

This paper provides some preliminary empirically based models which can be applied to predicting surface water quality in unsewered urban areas. These are based on the results of an extensive field monitoring programme undertaken in southwestern Sydney. Derived relations are then applied to the development of planning controls for unsewered urban areas.

UNSEWERED URBAN AREAS

In residential urban areas, the treatment and disposal of domestic sewage and wastewaters is a major concern for both public health and environmental quality. Unsewered urban areas differ from sewered urban areas in that by discharging contaminated wastestreams on-site, they are frequently net 'importers' of water and nutrients to the catchment. Through the disposal of sewage on-site, unsewered areas therefore have the potential to alter significantly catchment hydrology and hydro-chemistry in ways differing from traditionally observed impacts of sewered urban landscapes. This is notably the situation in areas which are serviced by reticulated water supplies (Figure 1). Typically, only excess stormwater runoff is evacuated from the catchment (Martens, 1996).

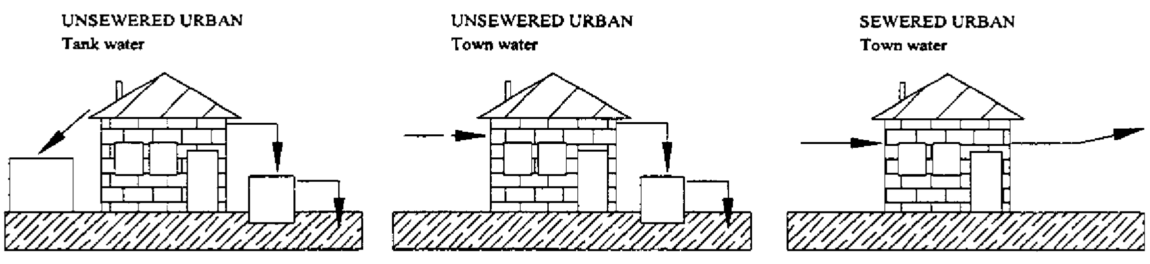

Figure 1: Types of urban areas indicating block water fluxes.

Unsewered areas may also differ from typical sewered areas in that land zonings are often rural-residential and allotment sizes are therefore correspondingly larger.

The potential for unsewered urban areas to contaminate both surface- and ground-water resources has long been recognised (Martens, 1996) and numerous studies have indicated that pollution from septic tanks and more recently Aerobic Wastewater Treatment Systems (AWTS) has the capacity to migrate from effluent application fields to local waters (Whelan *et al.*, 1979, Gerritse *et al.*, 1990, Martens & Warner, 1995, Kinhill Pty Ltd, 1997). In each of these studies there is a general concurrence that increases in the number of on-site wastewater management systems may lead to increasing risks of contamination of local surface- and ground-water resources. However, none of these works have yet quantified these relationships in order to develop urban planning controls which may improve urban water quality.

PLANNING CONTROLS FOR UNSEWERED URBAN AREAS

Presently, the risks to local surface- and ground-water resources incurred through on-site wastewater management practices are exacerbated by the prevalence of failing on-site treatment and disposal technologies. Numerous studies have reported that a high proportion of on-site systems may fail during their operational life (eg. Brisbane City Council, 1992; Kinhill Pty Ltd, 1997). In this case, failure includes both poor effluent quality from the treatment plant, and also failure of the disposal field. However, it is the failure of the disposal field which may lead to increased contaminant transport to local waters.

Urban planning controls which are currently implemented in relation to unsewered urban areas include one or more of the following:

1. minimum allotment sizes (eg. 2000-4000 m^2 for rural residential areas);
2. minimum effluent application sizes (eg 200 m^2 for AWTS irrigation areas);
3. minimum setback distances to water courses and groundwater supplies (eg. 100 m to streams); and
4. enforcement of accepted standards (eg. AS1547, 1994).

Most evident in these controls is that physical constraints on development are not typically based on site specific criteria. For example, minimum allotment sizes are generally arbitrarily derived or based on local knowledge and experience. They are not based on water quality objectives or land capability to accept effluent.

URBAN PHYSIOGRAPHY AND POLLUTANT RELATIONS: AN EXAMPLE FROM SOUTHWESTERN SYDNEY

Water Quality Monitoring Program

An urban runoff water quality monitoring programme was established during the early 1990's for 11 small unsewered urban and rural catchments. This aimed at identifying both the level of surface water contamination prevalent in unsewered urban areas, but also the importance of various environmental and urban controls on water quality. One of the key elements of interest was the influence of urban physiography on surface water quality.

The sampling programme was undertaken for approximately 18 months with surface water quality taken every 2 weeks from the downstream end of each study catchment. Although some 17 water quality parameters were analysed, only total nitrogen (TN), total phosphorus (TP) and Faecal Coliforms (FC) are reported here.

Study Area

The study area included monitoring conducted in 6 small villages in the southwest of Sydney (Figure 1). This included 11 sub-catchments which comprised of 3 catchments serviced by septic tanks and soil absorption systems, 3 by AWTS and irrigation areas, 2 mixed septic tank / AWTS areas, and 3 non-urban / rural sites which were used as control catchments.

Catchment physiography varied substantially between individual sub-catchments (Table 1), thereby allowing the influence of varying urban descriptors to be examined in relation to urban surface water quality. Catchments were deliberately chosen so that physiographic factors such as catchment area, housing density (ie. houses/ha), the percentage of sealed surfaces (Impervious %), the percentage of road cover (Road %), the curbage (m/ha), gradients and runoff coefficient (RC) varied between sites.

Table 1:
Summary of study catchment urban physiographic data.

Catchment	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**
Type	Septic	Septic	Mix	Mix	Control	AWTS	Septic	Control	Control	AWTS	AWTS
Area (ha)	11.92	3.10	8.14	21.56	3.17	15.90	16.68	2.21	25.81	39.67	4.51
Houses/ha	5.7	2.3	5.7	4.3	0	1.2	3.5	1.4	0	1.1	2.4
Impervious %	33.4	25.7	28.3	26.4	0	13.4	37.7	41.4	6.0	13.0	17.5
Road %	12.3	13.2	11.7	12.0	0	5.6	29.1	39.8	6.0	9.2	9.8
Curbage (m/ha)	248.3	377.8	371.9	388.1	0	134.3	324.7	584.5	70.3	117.4	112.5
Max. fall (m)	26	14	18	26	90	96	24	5	12	18	16
Max grade (%)	6.6	8.8	9.6	5.9	21.0	4.1	4.5	2.3	2.8	2.7	3.1
RC	0.51	0.47	0.46	0.45	0.55	0.47	0.50	0.34	0.34	0.33	0.43

Soils in each of the study catchments were reasonably similar, being derived from Wiannamatta Shales, being categorised as predominantly high clay content red podzolics, with low permeability (saturated hydraulic conductivity < 200 mm/day) and relatively high surface runoff coefficients. On-site systems in these clay areas are frequently found to fail and surface runoff of effluent is common (Martens, 1996).

Rainfall varies east-west with some 900 mm falling annually in the Appin area and approximately 680 mm falling at Oakdale and The Oaks.

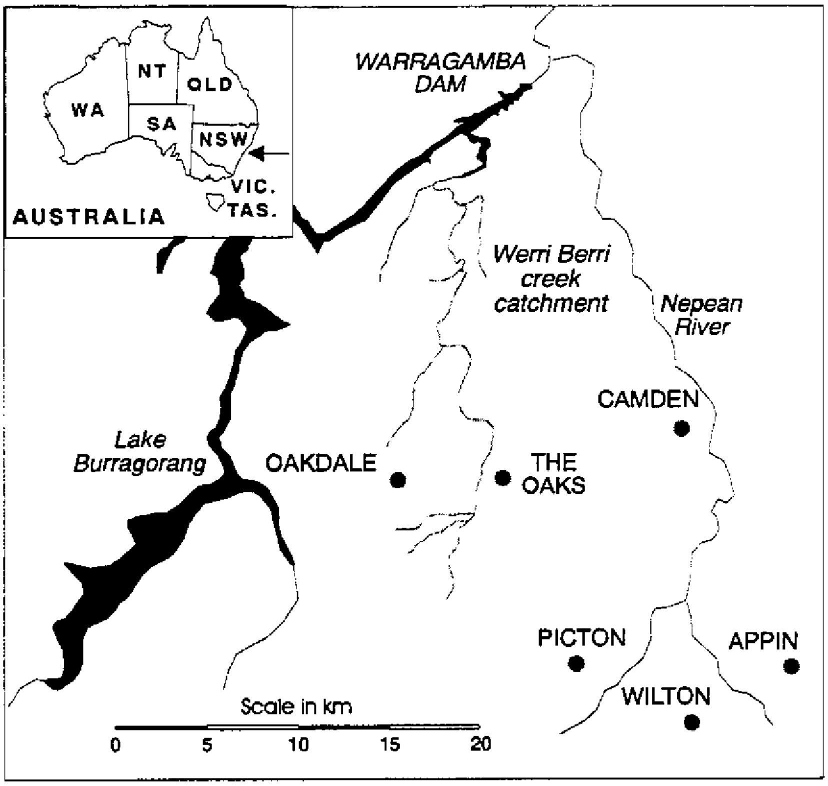

Figure 2: Location of study area catchments.

Curve Fitting and Model Estimation

Over the 18 month monitoring period, some 39 water samples were collected from each of the study catchments. These data were plotted against each of the urban physiographic parameters to determine the extent of relations between surface water quality in unsewered urban areas and the physical characteristics of the catchment. In all, 4 types of curve models were fitted to the data including:

1. linear;
2. exponential;
3. power; and
4. logarithmic.

Only the exponential model data are presented here due to space restrictions (Equation 1). Multi-variate analyses where also undertaken but are presented in Martens (1996).

$$C = e^{a} e^{bx} \tag{1}$$

where; C = surface water quality mg/L, for any given contaminant)
x = value of urban physiographic parameter (ie. housing density or sealed percentage)
a and b = model constants

Coefficients of determination were determined for each model fit so that relation significance could be understood.

Pollutant Concentration Relations

Results of regression analyses for pollutant concentrations (Table 2) indicate several significant relations between catchment physiography and surface water chemistry. Importantly, these relations vary during dry- and wet-weather, with dilution being most common during rainfall. This is not necessarily always the case, with faecal coliforms for example increasing during wet-weather above dry-weather levels in larger rural-residential style allotments. These relations are graphically illustrated in Figure 3 and Figure 4.

Table 2:
Results of regression analyses between pollutant concentration and housing density (H_D) and impervious % (IMP%). Coefficients of determination (R^2) are each significant ($p < 0.05$).

		Dry Weather			Wet Weather		
Water Quality Parameter	**Urban Parameter**	**a**	**b**	**R^2**	**a**	**b**	**R^2**
TN (mg/L)	H_D	0.355	0.34	0.468	-0.197	0.247	0.653
	IMP%	0.263	4.279	0.308	-0.29	3.231	0.462
TP (mg/L)	H_D	-1.629	0.398	0.275	-2.114	0.266	0.302
	IMP%	-2.279	7.463	0.403	-2.707	5.714	0.578
FC (CFU/100ml)	H_D	6.142	0.873	0.635	8.005	0.416	0.606
	IMP%	5.910	10.988	0.416	7.864	5.372	0.419

Note: H_D = Housing density (houses/ha), IMP% = Impervious catchment area (%).

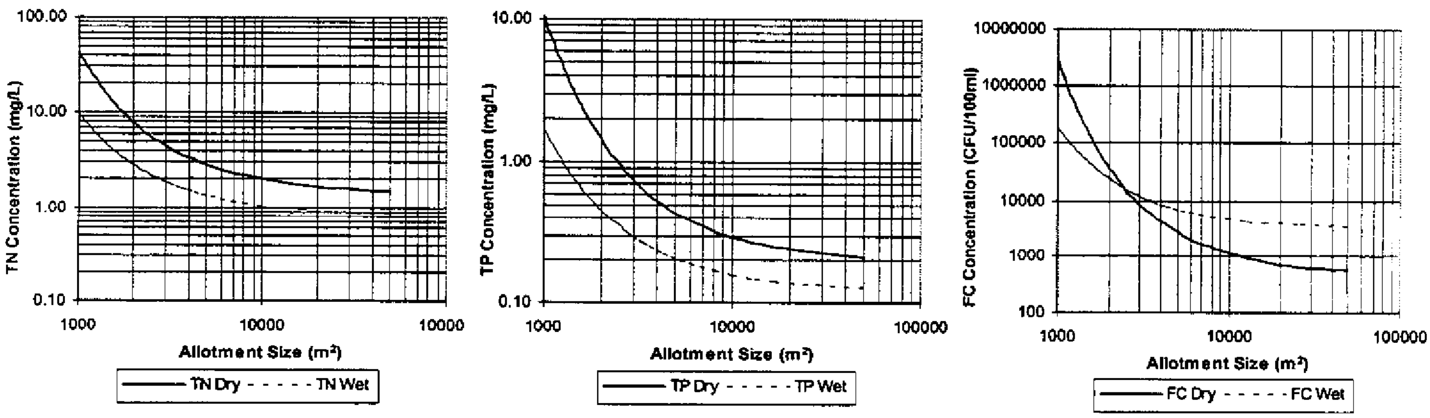

Figure 3: Log-log plots of contaminant concentration and housing density relations (TN = Total Nitrogen (mg/L), TP = Total Phosphorus (mg/L), FC = Faecal Coliforms (CFU/100ml)).

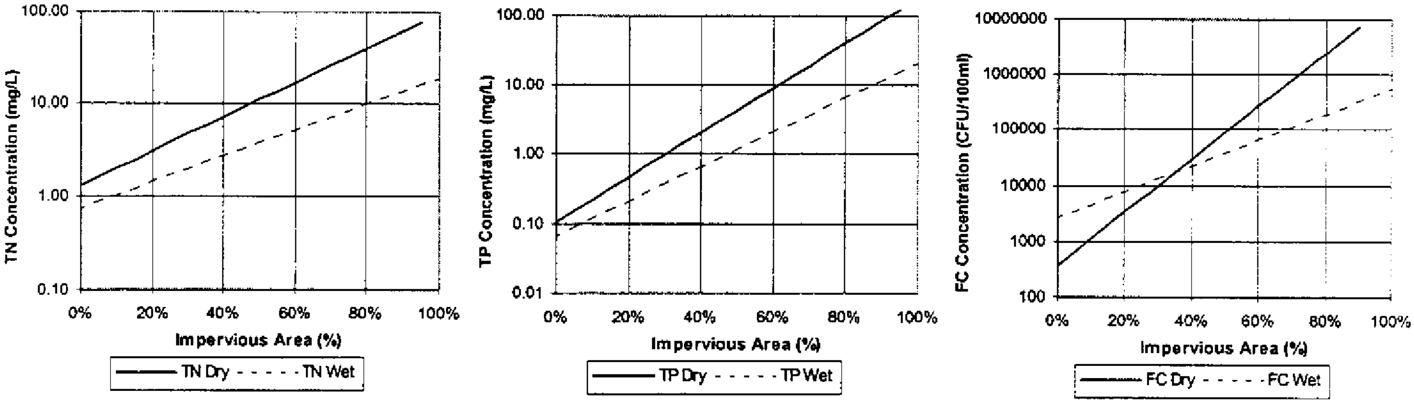

Figure 4: Seli-log plots of contaminant concentration and impervious area (%) relations (TN = Total Nitrogen (mg/L), TP = Total Phosphorus (mg/L), FC = Faecal Coliforms (CFU/100ml)).

Pollutant Load Relations

Results of regression analyses for unit pollutant loads (Table 3) also indicate several significant relations between catchment physiography and loads to local surface waters, although data have not been separated into dry- and wet-weather pollutant loads. Relations are graphically illustrated in Figure 5.

Table 3:
Results of regression analyses between unit pollutant loads and housing density (H_D) and impervious % (IMP%). Coefficients of determination (R^2) are each significant ($p < 0.05$).

	Parameter	a	b	R^2
TN (kg/ha/yr)	H_D	1.036	0.305	0.760
	IMP%	1.112	3.124	0.331
TP (kg/ha/yr)	H_D	-0.890	0.337	0.426
	IMP%	-1.308	5.733	0.510

Note: H_D = Housing density (houses/ha), IMP% = Impervious catchment area (%).

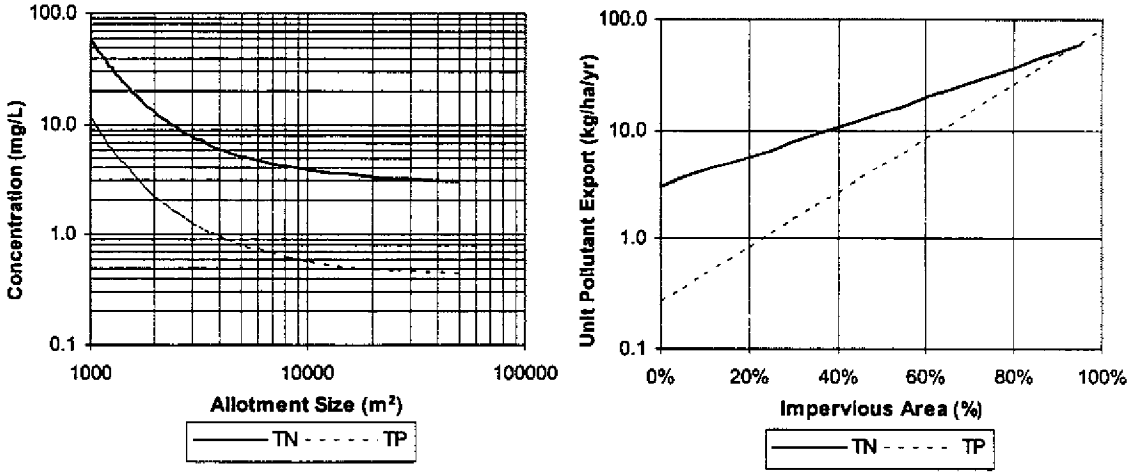

Figure 5: Unit contaminant loads (kg/ha/year) and catchment physiography relations (TN = Total Nitrogen (kg/ha/year), TP = Total Phosphorus (kg/ha/year)).

Application of Findings for Urban Planning Control

The results of urban physiographic regressions now make it possible to use physiography, for example H_D (housing density) as a generalised or bulk design parameter for unsewered urban areas based on specified receiving water quality criteria or objectives. Equation 1 may therefore be re-written so that H_D [or any other physiographic parameter] is the dependent variable subject to an input compliance surface-water contaminant concentration X_H (Equation 2).

$$H_D = \frac{\ln\left(\frac{X_H}{e^a}\right)}{b} \qquad \textbf{(2)}$$

where H_D = housing density (houses/ha)
X_H = Compliance contaminant concentration (mg/L)
a and b = model constants (obtained from Table 2 and Table 3)

Similarly the approach may be applied to unit contaminant loads and a specified maximum generation rate is exchanged for X_H. In this case, data from Table 3 would be substituted for that of Table 2.

CONCLUSIONS, LIMITATIONS AND FUTURE WORK

The preliminary work presented here represents a first attempt to quantitatively characterise the impact of urban physiographic factors such as housing density on surface water quality based on actual long-term monitoring data gathered in an Australian setting. The equations and models are not intended to be rigorous, excluding for example the impact of catchment grade, but rather to provide some guidance in an otherwise data-deficient environmental setting. The findings may thus be equally applied to other similar environments which receive annual rainfalls of 650 - 850 mm and are characterised by undulating soil landscapes maintaining impermeable high clay content soils with moderate to high runoff coefficients.

Importantly, several limitations to the approach outlined in this work exist which prevent its application to a larger set of environmental conditions. Most significantly, there has been no account of varying rainfall on contaminant concentration and unit load generation rates. These data have been gathered but are yet to be incorporated into the model.

Similarly, no account has been made for varying soil types [given that the study soil types are relatively uniform]. Most important here is the impact of soil permeability on the transference and division of contaminant wastestreams to either surface- or ground-waters. For example, in catchments with highly permeable soils, very little wastewater is likely to be distributed directly to local surface waters, but is more likely to be transferred without further renovation to ground-water (eg. Gerritse *et al.*, 1990).

Finally, it has already been shown by Martens (1996) that various of the physiographic characteristics are not independent. However, multivariate analyses already undertaken in Martens (1996) do not provide significant further insight into the analyses, most likely due the limited degrees of freedom which can be attributed to the small data set (11 catchments). The housing density parameter, H_D, however remains particularly useful in that it integrates a number of physiographic parameters such as IMP% and total pollutant load (ie. number of systems in the catchment). More work on multi-variate analyses is however warrented.

The author is currently in the process of addressing these issues through the development of a set of more complex models which take account of variability in rainfall and soil type.

ACKNOWLEDGMENTS

The financial assistance of Sydney Water and the NSW Department of Industry and Technology (DISTEC) are acknowledged. Sydney University's Department of Geography and its staff are acknowledged for their support and assistance during the early stages of this research when the majority of field work was conducted.

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THE PERFORMANCE OF ANAEROBIC SINGLE BAFFLED REACTOR (ASBR) WITH VARYING VFA CONCENTRATIONS

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ABSTRACT

The performance of two identically shaped laboratory scale anaerobic reactors was tested for volatile fatty acids (VFAs) removal. The control reactor simulated a septic tank, while the test reactor had a baffle separating the reactor into two compartments (inlet and outlet compartment) with an opening at the bottom. Both reactors were fed with an identical feed of acetic, propionic and butyric acids at concentrations of 2 mM.

Within less than 3 weeks, the anaerobic single baffled reactor (ASBR) showed that its performance was superior to the simulated septic tank. The ASBR degraded nearly 100% of VFAs, while the control could only degrade about 40%. Also at higher VFA concentrations the ASBR resulted in a much better effluent.

The results clearly indicated that the design of typical septic tanks did not lend itself to degradation of pollutants, but to solids settling. The introduction of a single baffle with an opening at the bottom (ASBR) improved the biomass feed contact time and provided better mixing between bacteria and substrate and gave a better effluent quality.

After gradually decreasing the hydraulic retention time, from 50 down to 6 hours, the build-up of acetic and propionic acids decreased the ASBR performance to about 55% VFA removal. At these retention times hardly any VFA degradation would be expected in septic tanks.

KEYWORDS

anaerobic reactor, ASBR, septic tank, acetic acid, propionic acid, and butyric acid, VFA

INTRODUCTION

The septic tank seems to be the most popular treatment system in developing countries, where some type of simple treatment is used. However, effluent from the septic tank may lead to contamination of groundwater (Matthew and Ho, 1995). Population concentration in third world countries causes a serious strain on urban infrastructure (Inamori, 1994). As a consequence, massive discharges of polluting effluent from extending urban housing cannot be avoided. In Jakarta for example, the Indonesian Environmental Bereau (BLH) reported that groundwater in 101 suburbs, tested through 300 shallow wells, was contaminated by E.coli (73%), organic compounds (35%) and detergent (90%) (Kompas, 1997). To prevent contamination of water resources, such as groundwater, a major improvement of the currently used septic tank treatment is needed. Applied technology can be used to solve this problem. It should not necessarily be, high-tech, but must have a high content of biotechnology knowledge (Niemczynowicz, 1995). Due to economic considerations small scale technologies should be considered. This will avoid the expensive infrastructure of a traditional sewerage system that uses scarce drinking water as a transport medium for pollutants.

Septic Tank

A septic tank is basically a self-contained installation for the pre-treatment of sewage. It takes the form of a settlement tank in which sewage is retained for a sufficient time for organic matter to undergo some anaerobic decomposition (Carter, 1995). It usually consists of one or two tanks for settling of solids with the overflow disposed via subsurface soil percolation. Even after passing through the soil the effluent from a septic tank does not usually meet the criteria for maintenance of acceptable ground water quality and hence needs further treatment (Ho and Mathew, 1993). Site suitability criteria should be applied for system location, and routine operational checks and maintenance activities should be conducted (Canter and Knox, 1985).

The septic tank functions to treat both black and gray water. In the USA typical sources of household wastewater, expressed on a percentage basis are toilet 22~ 45 %; laundry 4~ 26 %; bath 18~ 37 %; kitchen 6~ 13 %; and other 0~ 14 % (Canter & Knox, 1988). Because of the lower water consumption per head (less than 200 L per person per day) domestic wastewater streams are more concentrated in most developing countries.

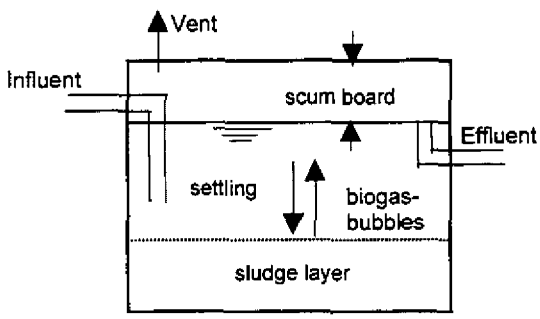

Figure 1. Longitudinal section of septic tank (van Haandel and Lettinga, 1994)

An important consideration for effluent quality from the septic tank is to prevent groundwater contamination. Based on a number of studies, the following represent typical effluent concentrations from septic tanks (Canter and Knox, 1988): 75 mg/l suspended solids, 140 mg/l BOD_5, 300 mg/l COD; 40 mg/l total nitrogen; and total phosphorus =15 mg/l. Studies on the efficiency of soil absorption systems have shown the typical concentration entering ground water: suspended solids = 18~53 mg/l; BOD5 = 28~84 mg/l; COD = 57~142 mg/l; ammonia nitrogen = 10~78 mg/l; and total phosphate = 6~9 mg/l. Bacteria, viruses, metals and inorganic contaminants are also found.

Small scale wastewater treatment

Sewerage system and centralized wastewater treatment plants based on end of pipe technology should not be considered as the only possible solution for sanitation (Otterpohl, *et al.*, 1997). Small scale solutions might become a new approach instead of end of pipe approach (Niemczynowicz, 1995). Small scale wastewater treatment can be defined as technologies which occupy small space for treating single house domestic wastewater up to a small community (around 5,000 to 50,000 people) (Saldinger, 1992). Houses, served by small scale wastewater treatment, do not have to be a part of a centralized sewerage system. Therefore, small scale technologies enable the installation in urban or suburban areas without any connection to the main sewer.

AIM OF THIS PAPER

The aim of this paper is to study the effect of a simple improvement to the septic tank system by inserting a baffle at the inlet to the tank resulting in better mixing of the biosolids with the incoming feed.

MATERIALS AND METHODS

Two identical reactors were set up and tested for VFA degradation. The reactors were physically similar in volume and shape. The first reactor had a baffle separating the reactor volume into two chambers. It allows the feed to flow through the sludge layer before leaving the reactor. The second reactor was a typical septic tank system. It was used as a control.

The reactors were operated at a hydraulic retention time (HRT) of 2 days, the normal HRT for a septic tank system. Temperature was set at 35°C as a base-line condition.

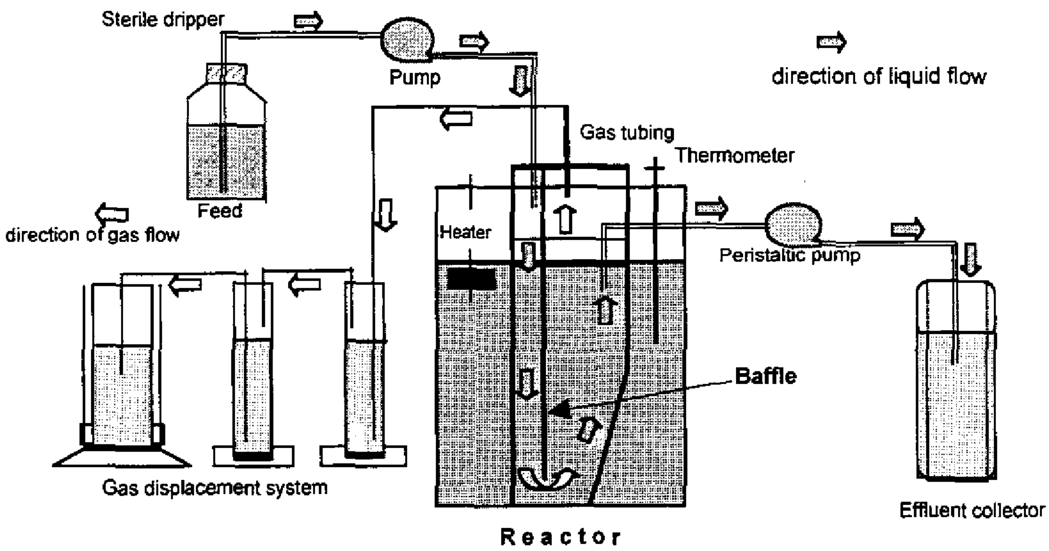

Figure 2. Schematic diagram of anaerobic single baffled reactor (ASBR).

Performance of both reactors, one with and the other without baffle was tested with various VFA concentrations. The reactors were started using biomass from an upflow anaerobic sludge blanket digester at Matilda Bay Brewery in Fremantle, Western Australia. That reactor had been run with an average load of 2000 mg/L BOD and HRT of 6 hour.

VFAs were used as feed to minimize variations in substrate characteristics. VFA concentrations were 2.0 up to 7.5 mM of acetic, propionic and butyric acids. The lowest concentration of VFAs used was 2 mM of all three acids. This is equal to approximately COD = 670 mg/l, which is similar to the level in domestic wastewater (Metcalf and Eddy, 1991). The highest VFAs concentration was 7.5 mM, which is more concentrated then black water.

The reactors were run in parallel on the same feed using a peristaltic feed pump for input and output. Gas outflow was connected to a displacement system and the biogas produced was read on a scaled gas measuring cylinder. CO2 in the biogas was removed by passing the biogas through concentrated solutions of NaOH. Then, the biogas volume measured was considered to consist only of methane gas. Previous studies had validated this assumption.

VFAs were analyzed using capillary gas chromatography with a flame ionization detector (VARIAN) and an automated injection system.

RESULTS AND DISCUSSIONS

Performance of ASBR on Treating Low Concentration of VFA

Acetic, propionic and butyric acids were degraded close to 100% in ASBR (figure 3, 4 and 5). This performance was achieved mainly because of the exposure of the bacteria to the feed. The inflowing feed was somewhat premixed with some bacteria in the first compartment of the baffled reactor, before it was inserted at the bottom of the second compartment. The second compartment worked like an upflow anaerobic sludge blanket (UASB). Good mixing between bacteria and feed and the sufficiently long contact between the two mean that the bacteria have enough time to consume all the feed. In the septic tank system, the degradation was 10%, 50% and 60% of acetic, propionic and butyric acids respectively (figure 3, 4, and 5). These figures clearly display the poor performance of an anaerobic tank with feed inlet and outlet at the reactor surface (i.e. short-circuiting the sludge) . Most of the feed was able to reach the reactor outflow without being exposed the bulk of the biomass. The high acetic acid levels were in part due to the production of new acetate from butyrate and propionate degradation (McInerney & Bryant, 1981). Also propionic was more difficult to remove than butyrate due to its less favorable redox potential making its oxidation more difficult. Thus the degradation pattern is in line with current mathematical and thermodynamic predictions of (Hoh, and Cord-Ruwisch, 1996).

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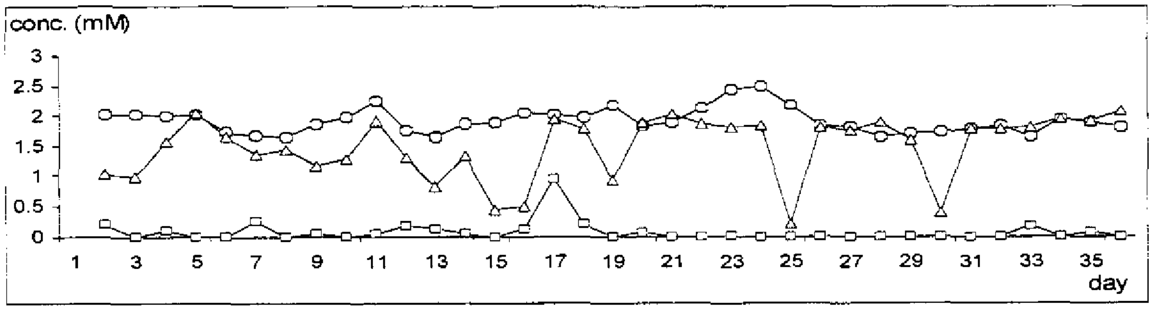

Figure 3. Concentration of acetic acid in the outflow of ASBR ( ), septic tank (Δ) compared with feed (o).HRT = 2 days. Temperature 35°C. Feed concentration 6 mM of acetate, propionate and butyrate.

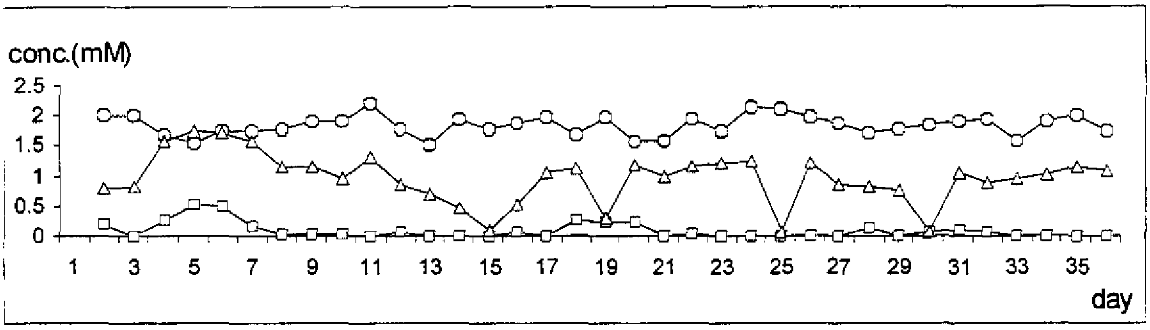

Figure 4. Concentration of propionic acid in the outflow of ASBR ( ), septic tank (Δ) compared with feed (o).HRT = 2 days. Temperature 30°C. Feed concentration 6 mM of acetate, propionate and butyrate.

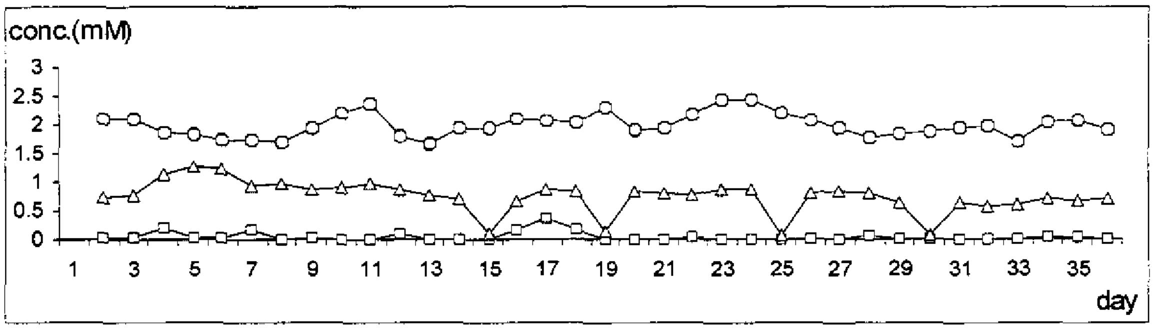

Figure 5. Concentration of butyric acid in the outflow of ASBR ( ), septic tank (Δ) compared with feed (o).HRT = 2 days. Temperature 35°C. Feed concentration 6 mM of acetate, propionate and butyrate.

Treatment of more concentrated waste

With an increased concentration of VFA (6 mM) the difference between both reactors became even more apparent. Once again, accumulation of acids occurred in the unbaffled septic tank system (Fig.3 to 7). Acetic acid was hardly degraded at all (figure 6).

Propionic and butyric acids were degraded 40% and 50% respectively, which was 10% less compared with previous degradation.

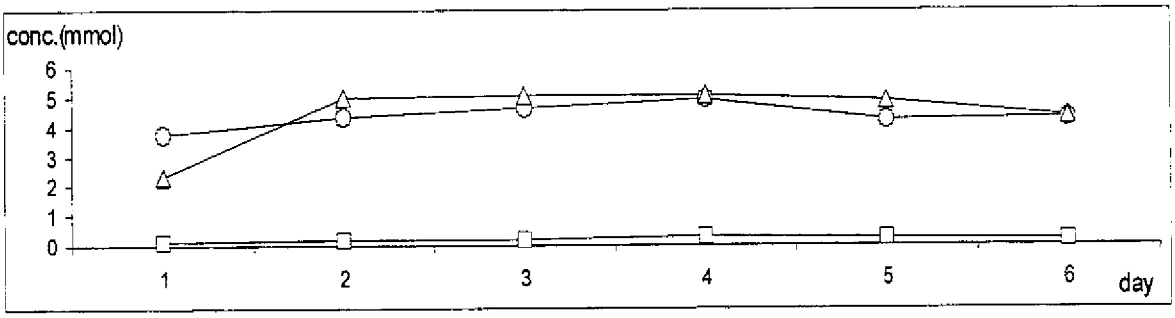

Figure 6. Concentration of acetic acid in the outflow of ASBR ( ), septic tank (Δ) compared with feed (o).HRT = 2 days. Temperature 35°C. Feed concentration 6 mM of each acetate, propionate and butyrate.

With a VFA concentration of 7.5 mM no change was observed in the concentration of acetic acid in the typical septic tank system (figure 7). Propionic and butyric were degraded by 30% to 40% respectively, again 10% less than previous case.

At this relatively high loading rate acetate started to accumulate to levels of up to 1.8 mM corresponding to about 75 % degradation. The other VFAs were still totally degraded in

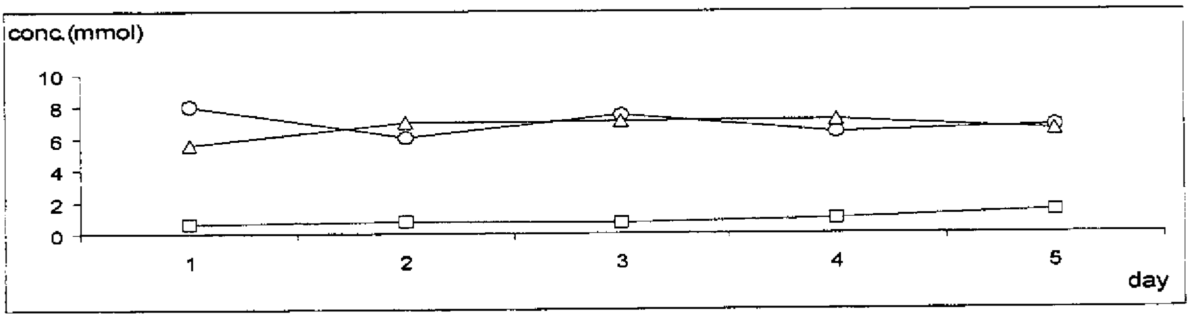

Figure 7. Concentration of acetic acid in the outflow of ASBR ( ), septic tank (Δ) compared with feed (o).HRT = 2 days. Temperature 35°C. Feed concentration 7.5 mM of each acetate, propionate and butyrate.

this reactor indicating that the drop in performance was not due to channeling of the feed through the sludge blanket but to biological limitation in terms of acetate conversion to methane and CO_2.

CONCLUSIONS

ASBR, which provides better mixing between feed and biomass, proved to be able to treat VFA concentrations up to 7.5 mM, while typical septic tank system showed very poor VFA removal even when treating fairly dilute feed. Further work with different waste waters and in pilot or full scale could test whether such a simple improvement in tank design can result in better outflow quality and a higher level of biomethanation, in particular for applications with increased waste concentrations such as in the developing countries.

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MICROBIAL POPULATION DYNAMICS IN WASTEWATERS FROM A POULTRY PROCESSING PLANT AND THE IMPACT OF AN AUTOTHERMAL THERMOPHILIC AERATED DIGESTOR. A PILOT STUDY.

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ABSTRACT

A high temperature Autothermal Thermophilic Aerated Digestor (ATAD) pilot plant was studied to quantify the nature and number of culturable organisms present. Incubations of wastewater and sludge was carried out at both 30°C and 60°C. It was found that resident microflora in the wastewater stream did not survive the ATAD process and that a highly motile, obligate aerobic, gram negative, sporulating bacilli dominated the ATAD effluent. The process which has operated at up to 70°C appears to be related to the presence of this organism. The high operating temperature creates a pasteurisation effect on resident species in the wastewater stream.. The heat produced in this process is potentially recoverable for reuse elsewhere. The stabilised and pasteurised sludge is useful for value adding products.

KEY WORDS

Aerated, Autothermal, Bacteria, Biomass Reduction, Culturable, Digestor, Pasteurisation, Sludge Stabilisation, Thermophilic.

INTRODUCTION AND BACKGROUND

Conventional wastewater treatment methods for the stabilisation and disinfection of bio-solids are generally expensive in both operator costs and the cost of chemical dosing. The resultant product is often unsuitable for sustainable environmental uses.

For food processing wastewater and their derived sludges, which typically exhibit an inherent high energy value in the solid particulate fraction, the use of thermophilic processes for treatment prior to reuse for land disposal or bioaugmentation is an emerging sustainable technology. The benefits of a thermophilic process is that at the right temperature it will both effectively pasteurise pathogens and stabilise the sludge. Sludges applicable for this process would include any unstable sludge, of high energy (kilojoules) content, derived from food processing or human sewage. There is the added potential of combining sludges or liquid wastes to enhance the process.

This study focuses on a poultry processing plant having a turnover of some 48,000 birds daily with a resultant wastewater stream of approximately 30,000 litres/hr over an eight hour day (Fennell . 97). The wastewater is derived from a number of sources which includes drainage from marinade tanks, wash down from the factory floor (feathers, meat, fat) and the hose down from the trucks and crates (faeces, soil etc). In keeping with good environmental practice a proportion of this water is recycled before being collected into a 250,000 litre holding tank where it is pumped to the treatment plant.

The existing wastewater treatment plant uses Dissolved Air Flotation (DAF) technology to treat the water to produce approximately 8000 L of sludge/day (Fennell 97). The remaining water is subsequently discharged to sewer under a trade waste licence. The sludge is stored in a 35,000 litres holding tank which is periodically drained by a licensed contractor for disposal. Preliminary investigations show that the DAF plant is capable of removing approximately 80% of the Biochemical Oxygen Demand (BOD) from the wastewater stream. (Fennell 97) The sludge normally produced is highly putrifiable and attracts complaints from local residents during the summer months. Sludge viscosity and composition is variable on a daily basis but contains 5-8% solids normally (Fennell 97). The ATAD process introduces air into the reactor for biological stabilisation of the organic substances.

The biology of the ATAD process resembles a high speed liquid composting system in that energy is released in the form of heat whilst at the same time organic solids are degraded. This generation of heat is carried out by aerobic thermotolerant or thermophilic organisms. Such processes normally operate between 50-60^0 C (Hamer 95).

Initial reports on the performance of the pilot plant indicated that within 24 hours of commissioning the plant with sludge from the DAF treatment plant temperature reached 60^0 C without preheating. The plant continues to operate autothermally at between 60-70^0 C and usually performs at 70^0 C. At this temperature genuine thermophilic process microbes as opposed to the thermotolerant mesophiles are able to develop and assert their potential (Mason et. al., 87). The mean cell resident time (MCRT) is approximately 5 days.

Biologically these observation raise some interesting questions such as: Which organisms are active in the process? Where are they coming from? Are they thermotolerant or thermophilic? Is the microbial community present in the waste water changing with time and how rapid are these changes?

This study examines the microbial consortia present in the wastewater stream and from the DAF treatment plant and the ATAD pilot plant. It examines the changes taking place in the microbial population dynamics throughout the digestion process.

AIM

To examine the relative abundance of culturable fungi, actinomycetes and bacterial populations in the process wastewater stream through an analysis of liquid in:

A. Holding Tank Wastewater (DAF influent)

B. DAF Sludge
C. ATAD Effluent (treated sludge) at both 30^0 C and 60^0 C.

METHOD

To support the growth of the three main groups of microorganisms (actinomycetes, fungi and bacteria) three different types of media were used. Sabourard Dextrose Agar (BBL) for the isolation and growth of fungi, Dermasel Selective Agar (Oxoid) for the isolation and growth of actinomycetes and Tryptic Soy Agar (BBL) for the isolation and growth of bacteria other than actinomycetes.

To quantify the numbers of microbes (actinomycetes, fungi and bacteria) in the wastewater of the influent of the DAF. plant, the sludge produced from the DAF. plant and the effluent from the ATAD process the following methodology was used.

1. 1ml of waste fluid from each source was diluted into 99 ml of sterile distilled water. Wastewater was collected from the holding tank, DAF sludge was collected from the front of the DAF tank and the ATAD effluent was collected directly from the pilot plant using 100ml sterile plastic containers.
2. Mixing was uniform for each step and comprised of a 3 minute agitation.
3. A further two 1/100 serial dilutions were done using the same protocol as described in the previous step. This resulted in a 10^{-2}, 10^{-4} and 10^{-6} dilutions.
4. Petri plates were labelled and dilutions were dispensed in such a way (either 1.0 ml or 0.1 ml) to give the following concentrations for each of the microbial groups as follows:-
   - Actinomycetes (10^{-3}, 10^{-4}, 10^{-5} and 10^{-6}.)
   - Fungi (10^{-2} , 10^{-3}, 10^{-4} and 10^{-5}.)
   - Bacteria (10^{-4}, 10^{-5}, 10^{-6} and 10^{-7}.)
5. Each plate was prepared in triplicate.
6. Using a HTL micro pipette, sterile disposable microtips and aseptic technique the dilute solutions were distributed to each of the Petri plates in accordance with the amount (either 1ml or 0.1ml) to arrive at the appropriate dilution.
7. 15ml of the Tryptic Soy Agar, at 50^0 C , was poured onto the solution in the Petri plate and the plate was swirled gently on the bench to mix the contents and then left for the agar to harden (ie, pour plate technique).
8. The same was done for the Sabourard Dextrose Agar and the Dermasel Selective agars.
9. The Petri dishes were sealed with paraffin film and taped together.
10. The Petri plates were incubated in an inverted position. Two plates from each set were incubated at 60^0 C and the third plate from each set incubated at 30^0 C. The 60^0 C plates were incubated in a humidified environment to prevent desiccation of the agar base for 48 hours.

RESULTS

Table 1 gives the absolute numbers of organisms found growing on each of the plates. Where colonies exceeded 250 the plate was labelled as TNTC (Too Numerous To Count).

		Wastewater		DAF sludge		ATAD sludge	
		60^0C	30^0C	60^0C	30^0C	60^0C	30^0C
Actinomycetes	$10.^{-3}$	0	8	0	0	0	0
	$10.^{-4}$	0	1	0	0	0	0
	$10.^{-5}$	0	0	0	0	0	0
	$10.^{-6}$	0	0	0	0	0	0
Fungi	$10.^{-2}$	0	TNTC	0	TNTC	0	0
	$10.^{-3}$	0	TNTC	0	TNTC	0	0
	$10.^{-4}$	0	19	0	43	0	0
	$10.^{-5}$	0	0	0	4	0	0
Bacteria	$10.^{-4}$	0	59	0	TNTC	TNTC	2
	$10.^{-5}$	0	12	1	TNTC	19	0
	$10.^{-6}$	0	1	0	43	0.5	0
	$10.^{-7}$	0	0	0	0	0	0

Table 1. Organisms culturable from Wastewater, DAF. and ATAD at 30^0C and 60^0C.

DISCUSSION

Population dynamics : Whilst there are limitations to the methods used in this experiment in that only culturable organisms were studied changes in microbial communities were evident. Throughout the ATAD process we have observed a significant change in the microbial consortia present with the loss of major taxa. The changing conditions generated by various steps in the waste water treatment process from the holding tank through DAF and ATAD are consistent with the findings of Bull and Slater (82) who state that changing end conditions induce changes in the microbial population composition. The observations are also consistent with ecological observations of populations in extreme environments.

Only a very few, if any, of the microorganisms which are present in relatively high numbers in the wastewater stream appear able to survive the treatment process through to the ATAD effluent. In a number of sampling events a number of these organisms were able to grow at 60^0C initially and as the ATAD plant is operating at a temperature greater than this it would appear that the organisms found growing at 30^0C in the ATAD influent are not the same organisms.

The DAF process: The DAF process serves to concentrate suspended solids, on which microbes adhere, into a sludge which then served as the source of the sample for this analysis.

Thus one would expect that the total number of microorganisms would be greater. Table one indicates that this is the case for both fungi and bacteria at 30^0C but not for actinomycetes . This was confirmed by a second series of replicate experiments aimed directly at this hypothesis. Actinomycetes are basically soil microbes with a hyphal growth morphology and it seems that these organisms whilst present in relatively small numbers in the wastewater are not being concentrated in the sludge produced by the DAF process or are being destroyed by the chemical pretreatment to enhance flocculation of the wastewater in the DAF plant. Overall the DAF sludge concentrated the microbial population by a factor of some thirty six (36) times at 30 ^{0}C. This is in direct contrast to an observed thousand (1000) fold decrease in the microbial population when the DAF sludge was incubated at 60^0 C.

Observations of plate growth of the bacteria found growing in the ATAD sludge indicate a highly mobile organism giving rise to cream spreading dendritic shaped colonies often merging into confluent growths. These colonies only grew on the surface of the agar despite the 'pour plate' technique being used which distributes the bacteria throughout the growth media.. The surface growth suggest an obligate aerobe. This was confirmed by growth of the organism on a stab subculture. Gram staining revealed a Gram negative sporulating bacillus species approximately 2.0 x 0.7 microns in size. A second organism, of similar size, which is secreting an antibiotics substances against the primary Bacillus species was also observed growing on the surface of the ATAD agar. It appeared non motile and only occupied about 5% of the total growth on the plate. The nature of this heat resistant antibiotic and its long term effects on ATAD resident populations is an avenue for further research.

Bacterial numbers: It is noted that the total number of bacteria growing in the ATAD process is approximately 1000 times less than the total number of bacteria found in the DAF sludge at 30^0C. However at 60^0C this trend is reversed. There are about 10 times more bacteria present in ATAD sludge at 60^0C then found growing in DAF at 60^0C.

The colony shape and gram stain of the main organism found growing at 60^0 C in DAF and ATAD are observed to be the same suggesting it is the same organism. However as no organism grew at greater concentrations in the DAF samples it could be argued that this organism was either a contaminant or a spore of a thermophile which is present in the wastewater in very low concentrations. If the Bacillus populations are derived from the germination of spores then the spores are entering the ATAD process either via the air being sparged into the sludge, as the air is not filtered in any way, or they are present as ubiquitous spores in the wastewater stream and as only small volumes are analysed there may be sampling errors.. If the latter is the case then our observations of the lack of growth of Bacillus at 60 ^{0}C in the wastewater and DAF fluids suggests that they are present only in very dilute concentrations.

The role of microbes in ATAD is commonly thought of as dependent upon a consortia of thermophilic organisms acting together to produce a desirable end product. Studies on single substrate ATAD processes identifies two types of heterotrophic microbial consortia. One type contains more than 90% of single primary substrate utiliser (Hamer 1990). Our observations support this type of consortia. The ATAD process observed here is dominated by either one or only a few organisms. Whilst it is fairly obvious that the ATAD influent is not a single substrate its exact chemical composition is not known and would be worth investigating further. The lack of any significant change in COD throughout the ATAD process however suggests that the MCRT of the sludge may correlate well with some aspect of sequential substrate biodegradation. The fraction of the substrates actually being utilised in the process could be a target for further research.

Bacillus species have been identified as the main organisms which grow in aerobic thermophilic processes (Sonnleiter and Fiechter, 1983) and furthermore it is members of this genera which produce thermotolerant protease exoenzymes (Mason et.al, 1992). The role of microbial exoenzymes in colloidal and particulate hydrolysis remains a matter for debate. But it appears to be a significant factor in such a process.

The fact that both Bacillus species observed appear to be obligate aerobes is significant in that at 60^0C one would expect the availability oxygen to the organisms would be-limiting. Whilst the operating temperatures may affect the solubility of oxygen the much higher oxygen diffusivity at this elevated temperature may balance out oxygen mass transfer into the system.(Hamer 1995). The aeration device used in this ATAD plant has enhanced the mass transfer of oxygen into solution in this treatment process.

Hydrolysis: The increased rate of hydrolysis within the ATAD process is not due to an increase in the total microbial number coming from the wastewater although there is an increase in Bacillus microbial species found growing at 60^0 C between the DAF and ATAD processes. This suggests that the process is regulated by enzyme activity from thermophilic or thermotolerant enzymes. Whether these hydrolytic enzymes are resident thermotolerant metabolic enzymes resulting from lysis of the bacteria present in the DAF sludge or are thermotolerant exoenzymes already present in the influent or whether they are being rapidly produced by a reduced thermotolerant or thermophilic bacterial population could be the subject of further investigation as the results could lead to developing parallel biotechnologies to enhance the ATAD process.

Whilst it was outside the scope of this study it is important to note that the autothermal processes occurring created temperatures which are equal to or greater than that at which domestic hot water is supplied. As the pilot plant was constructed from stainless steel it would be possible to further construct a heat exchange jacket to capture the heat generated by the process. Such a high calorific value of these sludges and make them attractive for use in the development of sustainable technologies for resource recovery and it is noted that Ube city in Japan is already employing methods to redirect such products into resource recovery streams.

Given a source of electrical power remote sites are able to gain a twofold advantage from the ATAD process. These are (a) a stabilisation of sludges and (b) the generation of useable heat. Alternatively if a source of electrical energy is not available it may be advantageous to consider converting the process to an anaerobic thermophilic digestion. This would eliminate the need for an aerator but require preheating of feeder sludge to maintain the anerobic thermophilic microbial populations . These bacteria are efficient methane producers and would make methane available for capture and reuse.

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ADVANCED TREATMENT

WATER USE MINIMISATION AND RESOURCE RECOVERY IN THE TEXTILES INDUSTRY

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ABSTRACT

In this paper, we examine the use of resources such as water, chemicals and energy by industry with a view to reducing both operating costs and future capital investment in waste treatment thus maximising profits. In most instances, optimal use of resources will involve a reduction in quantity of resources used as well as implementation (where possible) of resource recovery options. An additional benefit of such resource conservation and/or recovery is likely to be a reduction in the cost of waste management. Indeed, a lowered use of resources is generally considered to be the first (and often most important) step in a Waste Management Program (WMP).

The Centre for Water and Waste Technology at the University of New South Wales has been particularly involved in the examination of resource usage in textile finishing both in Australia and (in very preliminary investigations) in Indonesia. The Centre is also involved in the work program of the industrial wastewater R&D Joint Venture between the CRC for Waste Management And Pollution Control and the Korean Institute for Environmental Technology and Industry.

While general principles of resource conservation and reuse are presented, most attention is focussed on lessons arising from selected case studies. Presentation of both generic and industry specific issues arising from these case studies represent an important part of this paper.

KEYWORDS

Textiles, resource recovery, cleaner production, wastewater treatment

BACKGROUND TO THE TEXTILES INDUSTRY

Textiles Manufacturing

As we will use numerous examples from the textile industry, it is important that we define some terms of common use in this industry.

The major steps in the manufacture of a finished fabric product may be summarised as:

Raw Materials $\xrightarrow{\text{(spinning)}}$ Yarn $\xrightarrow{\text{(weaving or knitting)}}$ Greige Fabric $\xrightarrow{\text{(finishing or wet processing)}}$ Finished Fabric

The majority of textiles effluent comes from the wet or finishing processes, which involve cleaning, dyeing and finishing the cloth, although processes like sizing and singeing in the weaving stages do create low volume wastes that are high in BOD. Wet processes are carried out in a Finishing Plant which may be a part of an integrated mill that converts raw materials or yarn into finished fabric or part of a commission finishing houses that will buy greige fabrics and convert them into finished fabrics.

Textile mills can be divided into several categories depending on the type of fibre processed, machinery requirements and the process conditions. A major division can be made between woven and knitted fabric production because of the equipment needed. These categories can also be divided according to fibre type. We have recently been focusing on cotton and poly/cotton blend fabric finishing, although 100% polyester fabric finishing has also been considered.

The textiles industry in Australia is grouped into the Textiles Clothing and Footwear (TCF) Industries which are addressed as a whole by the Australian government on matters of policy. The Australian TCF manufacturing industries are small when compared with other countries. As a consequence, Australian manufacturers have to hold a large portion of niche markets to ensure the long runs necessary to justify investment.

Textiles manufacture in Australia enjoys less protection than clothing and footwear as it is intrinsically more competitive. This is due to its capital intensive nature, which hinders countries with lower wage costs from gaining significant production cost advantage over Australia (as opposed, for example, to clothing manufacture). Strategies for survival of the Australian textile industry include:

- flexible highly skilled work force
- non-price product focus
- increased uptake of technology
- product design and marketing perfection
- commitment to quality
- increased export penetration
- strong supplier networks
- short production time
- responsiveness to market demand
- world best practice techniques

Thus, with regard to optimisation of use of resources (including recycling), Australian industry generally has an interest in world best practice and new technology, though the "tight" profit margins necessitate strong economic justification for a process change or introduction of a new technology. The tariff reduction program has been slowed recently, which may increase capital investment in the sector; in the short term, however, confidence is low and long term improvements that demand large capital outlays are not considered.

GENERAL PRINCIPLES OF RESOURCE CONSERVATION AND RECOVERY

Recently, Best Management Practice (BMP) Guides for the textiles industry have been published in the UK and the USA (ETBPP, 1996 and USEPA, 1996 respectively). In both cases, stress is placed upon the minimisation of water, chemicals and energy usage and on the possibilities for resource recovery. Both Guides emphasise the importance of instilling an awareness of resource conservation in staff. It is the plant operators who must notice inefficiencies like baths overflowing and take corrective action. Incentives for resource minimisation, production and quality maximisation help to reinforce these ideas. Identification of a "champion" to progress resource conservation and recovery opportunities is generally considered essential.

Preliminary Steps in Resource Conservation

It is best to formulate initially a ***resource conservation or waste minimisation program***, outlining milestones and duration of each phase of the program. The schedule could cover the first year of the project initially and could be extended as the need arose. The improvements that are implemented as a part of the program need to be monitored to maintain the initial level of benefit. Theoretically the schedule is only completed when resource conservation practices are totally absorbed into employee and management practice and there is no need for the co-ordinator or champion role.

A ***water, chemical and effluent audit*** will allow team members to visualise where chemicals that become apart of the waste stream are added in the process.

The purpose of the water, effluent, chemical and energy audit is to:

- allow the identification of the most valuable waste stream. The effluent should be assessed as the sum of all chemicals being discharged at new chemical costs and heat energy that is unrecovered at total generation cost, so the cost savings of effluent reduction can be highlighted. Expensive effluent streams can then be targeted for effluent reduction.
- allow the effluent characteristics to be compared to effluent characteristics published in BMP Guides so the company can gauge how efficient the process techniques are. Comparison of the litres of water consumed per kg of fabric to values found in best management practice literature can be taken as a rough guide of the efficiency of resource use. The survey may also reveal if there are any major leaks that have been overlooked.
- evaluate the individual effluent streams in terms of their recyclability . If streams can be treated before mixing, the clean streams can be reused directly and the dirty streams remain more concentrated permitting cheaper and more efficient treatment. Variation of effluent characteristics must also be gauged so that the treatment step(s) have the capacity to treat the effluent at all times to a recyclable quality. The production of effluent that is not an acceptable standard to be recycled needs to be correlated with unit operations and recipes in the plant. Then unacceptable effluent generation can be predicted and streams diverted from the waste treatment plant. To assess completely the variations in quality of individual effluent streams, a monitoring period of at least three months is required.

A survey to confirm ***material flow information*** is required to perform an adequate cost analysis of any waste minimisation proposals that may be put forward.

The general philosophy is to invest preferentially in process optimisation and waste minimisation rather than end of pipe effluent treatment. To segregate effluent and recycle certain streams to unit operations that are compatible with the waste quality is preferred to treating and reusing the bulk effluent. A brain storming session should be held to generate ideas for minimising specific waste areas.

Actions to Reduce Resource Usage

If the collection of information for the resource survey was difficult, it was probably due to the lack of a ***resource inventory*** system. The introduction and maintenance of a computerised chemical accounting system will allow an audit to be conducted with greater ease in the future and is vital to the success of the resource conservation program. An advanced system may track the production and disposal of different grades of water (Eg heated or non-heated; clean or dirty process water).

With production pressures increasing, simple good house keeping measures tend to be ignored. Issues that may need attention include:

- Prevention of excess chemical make-up,
- Prevention /avoidance of spills and overflowing baths,
- General maintenance to eradicate leaks and keep equipment performance up to specification. Maintenance of pipes and flow meters,
- Water efficient cleaning methods; minimum hose use for washing equipment and floors,; use of wastewater for equipment wash down
- Maintenance of dripping taps and valves,
- Preventative maintenance.

Raw materials can be replaced by ***less polluting chemicals*** which will decrease effluent pollution load and hence the cost of effluent disposal, or will increase the reusability of individual effluent streams.

Optimisation of utilities could be addressed after/during the preparation section analysis. Issues to address include:

- ensuring the mains water pressure is stable, so water consumption of equipment does not deviate from an optimised value.
- performing an energy audit of the boiler house to make sure equipment is running efficiently.
- directly recycling water used for heating and cooling back to the heating and cooling process if possible or if it is not possible, as rinse water in a suitable process.
- adding heat recovery units where viable.

Optimisation of temperature, water flow, cloth speed, water levels and tank sizes of the existing processes will prevent unnecessary resource loss. Optimisation often requires the introduction of better control schemes for temperature, water flow and chemical dosing. This is a key area for resource reduction.

A decrease in the consumption of ***water*** will often translate to savings in ***chemicals and energy***. Water usage will probably be focused on during this stage of the WMP.

Measures to improve ***efficiency*** include:

- the use of more efficient washers especially continuous counter current washers,
- introducing most of the water to the process directly onto the fabric via a double showerhead (one on either side of the cloth).
- optimising rinse temperature,
- optimising water level in the rinse bath,
- decreasing drag out by increasing compression of nip rollers,
- introducing drying cans or vacuum slots after the rinse.

Because a water saving will result in the increase of contaminants in the effluent, waste disposal costs may increase if regulations specify maximum concentration limits. In the case of chemical recovery, however, an increased concentration of the recoverable compound represents lower capital and operation cost outlays for a recovery plant that will eventually be reducing overall operation costs.

Investment in new plant equipment can often save resources in the long term. For example multifunction equipment will reduce the number of individual process steps, as with combined desize scour or scour bleach or desize scour bleach in oxidative desize. Oxidative desize is a process which utilises H_2O_2 or persulphate to desize and scouring fabrics. A wide range of sizes can be removed and the COD/BOD of the effluent is reduced.

Equipment with lower specific water consumption could also be attractive.

Resource Recovery Actions

To maximise resource recovery waste streams should be ***segregated***. Plant engineers may feel that keeping effluent streams separate is inappropriate because the effluent piping system is complex (and sometimes undocumented). Even though an end of pipe solution seems simpler, more resources are wasted. Attempts to recycle the combined effluent usually fail. In most cases, expensive treatment steps are employed to produce water that is not reusable, because of its high unknown dissolved solids content. It is more advantageous to take individual streams with largely known constituents and recycle the effluent with minimal/no treatment steps.

Reduction in the number of fresh water feed points, by replacing them with ***recirculated water***, will decrease water consumption considerably. Wash water from cleaner processes can be reused as wash water from another process, and more concentrated waste streams can also be utilised if they are pre-treated. Table 1 outlines the water quality requirements for the various textile processes, indicating where recycled water could be used.

Effluent Management

A number of steps may be introduced to ensure better control of effluents including:
- Dry clean up of chemical spills
- Preliminary filtration/screening of effluent to remove fibre.
- Regulation of the effluent flow with an effluent storage.

After the waste volume has been minimised, and as many streams have been recycled and reused as possible a suitable ***effluent treatment process*** should be chosen from available technologies. These technologies have been extensively documented elsewhere (e.g. Barclay, 1996). Options for introduction of new technologies for effluent treatment (and potential recycle) should be assessed but this assessment must invariably be done on a factory by factory basis.

Table 1. Minimum Requirements for Input Water to Various Textile Processes

Process	Requirements
Desize Pad	If an enzyme desize is used then the water into the process must not affect enzyme reactivity.
Desize Wash	Because the fabric will always be scoured after, a poor quality water can be used. Scour effluent may be appropriate.
Scour Pad	High quality water only for chemical makeup.
Scour Wash	Towns supply water is of too high a quality for use as wash water. Slightly acidic water would be suitable as long as the effluent is alkaline. Mercerise or bleach effluent would be appropriate.
Mercerise Pad	High quality water only for chemical makeup.
Mercerise Wash	A relatively clean input stream is required. Bleach effluent could be used but cotton wax and lint must be removed so fouling in the caustic recovery plant is avoided.
Bleach Pad	High quality water only for chemical makeup.
Bleach Wash	Wash water must be colour free, with low salts and alkalinity and no oils.
Dye make-up	High quality water only for chemical makeup.
Dye	No salts, oils or colour that bonds to fabric.
Print	Solid free, low corrosive, colour free and low salt.
Finishing	High quality water for chemical make-up.

KEY

Cleanest input water	Medium quality input water	Dirtiest input water

Implementation and monitoring

Waste minimisation proposals are implemented according to their technical, economical and environmental feasibility. . Simple, low cost plans with short pay back periods should be implemented first to secure early improvements in cash flow and encourage/fund further work.

A procedure must be installed to monitor waste production and resource use to assure continued benefit from the projects that are implemented. Efforts to communicate the savings achieved in newsletters, bulletins and posters should be made to keep operators interested in resource conservation and reuse, and waste minimisation issues.

CASE STUDIES

The results of examination of resource usage at four textile factories are presented below. Two of these factories are in Australia (A&B) and two in Indonesia (C&D). Most detail is presented on Factory A since this textile mill has been extensively investigated and only preliminary inspections of the Indonesian factories have been made. Factory identities are, in all cases, considered confidential.

Company A (Australia)

Products and operations

Company A produces finished woven fabric, which is predominantly cotton, for furnishing and clothing products,. principally sold to a local market. ISO9002 standards have been implemented, which has opened up further markets. In the past few years, the company has made significant losses so funding for capital works is not present.

The company has facilities for weaving, fabric preparation (desize, mercerising, scouring, bleaching), dyeing (jet and continuous) and finishing (sanforising, water and fire proofing). Dyes used include, reactive, acid, disperse, direct and some azo dyes. Chemical finishes include water proofing, stain resistance, fire proofing, moth proofing and easy care. The equipment used ranges from inefficient winch dyeing machines to state of the art quality control instrumentation with automatic data logging capabilities. A wide variety of fabrics is produced, from a range of fibre and fibre blends.

Wastewater character (and problems)

The wide variety of products means the properties of the waste vary dramatically. The effluent pH is around 12 which is unacceptably high for the local waste authority. The authority has required that the textile mill effluent pH be reduced to 11; ie the concentration of NaOH must be more than halved.

Solutions considered

The company had only considered end-of pipe solutions to combat their pH problem such as

- Liquid CO_2 neutralisation of effluent alkalinity;
- Flue gas neutralisation of effluent alkalinity;
- Construction of an aerated flow equalisation dam that will hold effluent production for at least one day.

Waste minimisation strategy

The recommended Waste minimisation strategy redirected resources spent on end of pipe solutions to target high NaOH consuming processes. An overall audit of the plant was not appropriate, because of limited resources and the lack of process flow data available. The waste minimisation program was scaled down, to address the company's situation, and focused on the mercerising unit in the preparation section.

The waste problems could have been addressed in one action by reorganising the process sequence in the preparation section. As this would have represented a large capital outlay, unacceptable to the company, a minimalist approach, which involved increasing the efficiency of the existing mercerising operation was adopted. This encompassed everything from general maintenance to fixing leaks and clearing ancillary piping, to the installation of better cleaning systems for process streams, which would prevent fouling in the future.

Potential savings

$200,000/year is spent in caustic (not including water and waste charges) for the merceriser alone (not the entire preparation section). A significant portion of this expense could be saved.

Company B (Australia)

Products and operations

A wide range of cotton and cotton synthetic blend knit fabric is produced primarily in tubular form although open width fabric is sometimes made. Knitting, dyeing and finishing are performed on the same site.

Previously, company B supplied manufacturers of large volume low value-added consumer necessities. With the restructuring of the Australian TCF industry in the 1980's and 1990's, however, company B now produces for clothing companies in the high value added quality markets.

Fabric preparation requires only bleaching; no scouring is necessary because of the nature of the cotton received from Australian cotton growers. Pad batch machines are used for dyeing cotton and washing off is done in a winch. Softeners are added in the last stage of this process to almost all fabrics. Silicone is used for a very soft "satin touch" finish. Resin treatments for wrinkle resistance are also performed.

Waste water character (and problems)

The overall preparation effluent is of low pH which neutralises the predominantly alkaline effluent from the dye house to pH 10. Finishing processes produce little effluent but the effluent is usually very concentrated.

Approximately 390ML of effluent are discharged to sewer per year by Company B. Trade waste restrictions on effluent flow rates and composition are normally met. Grease and oil content of the effluent causes occasional problems but is not a significant cost to the company

Solutions considered

Company B's main concern is the escalating price of water. The present price of $AUD1.5/KL of water usage (including effluent discharge) is predicted to increase to $AUD2/KL by the year 2000. This means that the water bill will rise to $800,000 if they maintain the same production rate as today. With additional production expansion this figure could be significantly higher.

To combat the problem the company has conducted a leak audit and made preliminary investigations into water reuse of cleaner wastewater from the dyehouse. However, the company was not satisfied that the quality of water produced could be consistently high enough to have a negligible impact on the fabric.

Waste minimisation strategy

The waste minimisation strategy adopted should be an overall program (as outlined in section 3). Company practices should be scrutinised and a chemical accounting system further developed. Reuse of any clean effluent streams by recycling it to appropriate unit operations should be investigated. If the effluent did not meet effluent criteria, an effluent treatment step of adequate capacity would be required. Company B is an example of a textiles mill that would benefit from affordable membrane technologies developed for the reuse of clean waste streams.

Potential savings

The potential savings in water are gauged by comparing best practice water consumption figures (L of water consumed per kg of cloth produced) for the industry to the company's water usage. Current best practice data is very hard to find however and will vary enormously from country to country. The water consumption by various knit fabric finishers is shown in Table 2; taken from US EPA's Manual for Best Management Practices for Pollution Prevention in the Textile Industry (USEPA, 1996). Company B has a consumption of 95L/kg, which appears to be quite close to a middle water consumption but with current practices and technologies it seems reasonable that this consumption could be further reduced.

Water Consumption (L/kg)		
Range	Simple	Complex
Max.	275.2	276.9
Med.	78.4	86.7
Min.	12.5	10.8

Table 2. Water consumption for different knitting mills in the US.

If water consumption at Company B could be reduced to the minimum water consumption level of 20L/kg of fabric, (or best environmental practice level), the annual water bill would be reduced from $600,000/year to $126,000/year. This saving in water would result in a important savings in energy and chemicals as well.

Company C (Indonesia)

More than half of the fabric produced is 100% polyester for Middle East markets. Acid burnout cotton is also a major product. The processing details are summarised in the following notes:

- Fabric to H_2SO_4 bath in 1500 m production run, at 20 m/min.
- Allowed to react for 15 min. in J-box
- Water added in four places to the washing boxes; last washbox at 90°C
- Fabric passed over drying cans
- Acidic washwater (pH<1, 5-48 g/L H_2SO_4, 50°C) neutralised with $Ca(OH)_2$
- Alkaline (pH 11+) washwater plus $Fe(OH)_3$ and polymer to clarifier
- Clarified wastewater to river (pH 5.5-9.5, no foam, a little colour)
- Lime sludge from clarifier centrifuged and solids trucked to disposal.

A possible options for resource conservation and/or recycle would be to install a vacuum slot between the acid bath/J-box and the washer train to allow acid recycle; research may be needed on acid/dye separation to avoid dye contamination. Even if the acid is not recycled, fabric washing would be faster and neutralising the acid before dilution would be cheaper as well as saving on land for dams.

The change to counter-current fabric washing would require two pumps required but would result in major reduction in washwater used.

Company D (Indonesia)

The factory produces suiting material in cotton polyester/cotton and polyester/viscose blends, for domestic and export markets.

The processing details are summarised in the following notes:

- Approximately 91 L of water/kg fabric is used in the preparation step;
- Wastewater leaves the factory at pH 11; H_2SO_4 is added reducing the pH 6 - 7 (a lot of acid is used in this step!);
- Activated sludge plant is used for effluent treatment;
- Wastewaters from various stages of the plant are not cooled (ex scour and bleach - up to 70°C; at dyeing range 60°C; in to WWT at 40°C; at activated sludge ~ 30°C)

Options for resource conservation and recovery could be, subject to detailed study:

- Significant scope for savings in water consumption exist. For example, 91 L water/kg fabric in preparation is much too high - could go to 9 L/kg or less. Total water should be under 90 L/kg;
- The possibility exists of improved control of counter-current flow through installation of a rotameter;
- Caustic recovery from mercerising unit may be possible and may negate the need for acid addition to effluent;
- Heat recovery from effluent - 3 months payback on heat exchange may be possible
- Reactive dyes are widely used by Company D but biological treatment won't take much colour out of the effluent. If flow rate through the activated sludge plant is reduced, longer residence time and hence improved treatment of desize and scour effluents will result

ONGOING WORK

Perhaps the most important generic issue arising from these four case studies is that every factory is different and will require detailed analysis in order to develop options for cost saving through resource conservation and recovery. Usually larger savings can be made in the preparation section because the more simple process chemistry permits chemical recovery. In the dyeing section water is recoverable but dyes or other chemicals are not cost effectively recoverable with the available technology. Simplifying the process chemistry of the process will make wastewater more reusable.

The next stage of this program is to conduct detailed studies with the textile mills described above and help implement waste minimisation programs.

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The UNEP International Environmental Technology Centre (IETC)

The International Environmental Technology Centre (IETC) was established by the United Nations Environment Programme (UNEP) in April 1994. It has offices at two locations in Japan - Osaka City and Kusatsu, Shiga Prefecture.

The Centre's main function is to promote the application of Environmentally Sound Technologies (ESTs) in developing countries and countries with economies in transition. IETC pays specific attention to urban problems, such as sewage, air pollution, solid waste, noise, and to the management of freshwater basins.

IETC is supported in its operations by two Japanese foundations: The Global Environment Centre Foundation (GEC), which is based in Osaka and handles urban environmental problems; and the International Lake Environment Committee Foundation (ILEC), which is located in Shiga Prefecture and contributes accumulated knowledge on sustainable management of freshwater resources.

IETC's mandate is based on Agenda 21, which came out of the UNCED process. Consequently IETC pursues a result-oriented work plan revolving around three issues, namely: (1) Improving access to information on ESTs; (2) Fostering technology cooperation, partnerships, adoption and use; and (3) Building endogenous capacity.

IETC has secured specific results that have established it as a Centre of Excellence in its areas of speciality. Its products include: an overview on existing information sources for ESTs; a database of information on ESTs; a regular newsletter, a technical publication series and other media materials creating public awareness and disseminating information on ESTs; Local Agenda 21 documents developed for selected cities in collaboration with the UNCHS (Habitat)/UNEP Sustainable Cities Programme (SCP); Action Plans for sustainable management of selected lake/reservoir basins; training needs assessment surveys in the field of decision-making on technology transfer and management of ESTs; design and implementation of pilot training programmes for adoption, application and operation of ESTs; training materials for technology management of large cities and freshwater basins; and others.

The Centre coordinates its activities with substantive organisations within the UN system. IETC also seeks partnerships with international and bilateral finance institutions, technical assistance organisations, the private, academic and non-governmental sectors, foundations and corporations.

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