The Performance of Full-scale Waste Stabilisation Ponds Treating Saline Wastewater with Particular Reference to Bacteriophage as a Hydraulic Tracer

by

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SUMMARY

1. This thesis describes the establishment of a comprehensive monitoring programme instituted and operated by the author on the sewage treatment system on Grand Cayman, Cayman Islands on behalf of the Water Authority of the Cayman Islands.

2. The monitoring programme was established in 1988 and has run for more than seven years. It includes the essential chemical, physical and biological parameters required to monitor wastewater lagoon performance.

3. The programme was established with the practical aim of ensuring that the final effluent met microbiological and chemical guidelines which would permit the reuse of effluent for irrigation.

4. The study revealed that high saline groundwater intrusion into the sewers is such that the effluent cannot be reused at present. Repairs to the sewerage system reduced the salinity and flow in the final two years of the study but the salinity is still above the level which would permit healthy plant growth. Throughout much of the life of the system the groundwater intrusion also caused the hydraulic loading on the treatment plant to be at or above the maximum design flow.

5. The sewage treatment system is dominated by an unbalanced sulphur cycle resulting in high levels of hydrogen sulphide being produced largely from seawater-derived sulphate.

6. Since saltwater and salinization are increasingly important considerations in water and wastewater treatment, and have been shown to profoundly influence aerobic pond performance, more detailed insight into the mechanisms was required. These mechanisms are reviewed and investigated.

7. The diversity of phytoplankton and zooplankton normally associated with aerobic maturation ponds was greatly reduced by hydrogen sulphide generated in the facultative ponds. This aggravated the hydraulic overload and resulted in poor performance with respect to faecal coliform removal in the maturation ponds.

8. For more than 4 years the final effluent quality was not adequate for reuse because the anticipated faecal coliform removal of 99.99% was not achieved, and thus was
inadequate for reuse (>1000 faecal coliforms cfu/100 ml). Consequently, a fundamental study of lagoon hydraulic retention time was undertaken to validate and assess whether the design specifications and operating conditions are appropriate for the faecal coliform reductions required.

9. *Serratia marcescens* bacteriophage survival *in vitro* in raw sewage and effluent from the facultative and maturation pond was studied to investigate and compare survival characteristics with faecal coliform bacteria and with viruses and also to assess the prospects for using phage as a tracer of dispersion and retention time.

10. Phage death was shown to be affected primarily by high pH with rapid death above pH 9 associated with algal photosynthesis. Survival times in other conditions were adequate to permit *Serratia* bacteriophage to be used as a full-scale, hydraulic tracer.

11. Primary facultative pond 1.1 was dosed, *in vivo*, with a *Serratia marcescens* bacteriophage suspension. In-pond analysis of dispersion was conducted in addition to measurements of the phage in the effluents of each pond in the treatment system. The results were used to determine the dispersion characteristics of flow in the ponds and pond outlets samples were used to calculate the mean retention time.

12. Short-circuiting was identified experimentally and a numerical model using physical data of the facultative pond and climatic data, was subsequently applied. The retention time resulting from the application of this model and that shown from the tracer study were more closely correlated than either of them with the design retention time. The factors of wind direction and windspeed were identified as having a potentially major, negative influence on the hydraulic regime of the ponds when the wind varied from the east to north.

13. A series of engineering design change recommendations is presented which are intended to improve pond performance by improving the hydraulics of the system.

14. A comparison of the design performance in terms of pathogen indicator removals and biochemical reduction was carried out using data obtained from the routine monitoring programme and specialised intensive studies. There was good agreement between the level of short-circuiting defined experimentally and the actual removal of faecal indicators in the facultative ponds. This confirmed that retention time is the most important determinand controlling indicator reduction.
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Thank you, to all those who helped to make this happen and yes, I am finally through!

... At the far end of your tether
And your thoughts won't fit together
So you sleep light or whatever
And the night goes on forever

Then your mind changes like the weather
You're in need of Doctor Tarr and Professor Fether....

Tales of Mystery & Imagination Edgar Allan Poe,
The Alan Parsons Project

This Thesis is Dedicated to my Mother, Mrs Veta Frederick.
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</tr>
</thead>
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<tr>
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</tr>
<tr>
<td>Fig 10.25</td>
<td><em>Serratia marcescens</em> bacteriophage in outlet of pond 2.1</td>
</tr>
<tr>
<td>Fig 10.26</td>
<td><em>Serratia marcescens</em> bacteriophage in outlet of pond 2.2</td>
</tr>
<tr>
<td>Fig 10.27</td>
<td><em>Serratia marcescens</em> bacteriophage in outlet of pond 1.2</td>
</tr>
</tbody>
</table>
CHAPTER 1

1.0 INTRODUCTION

1.1 General Description of Study Area

The Cayman Islands, a British Crown Colony, consists of three flat limestone islands located in the northwest Caribbean, latitude 19°20'N and longitude 81°20'W, 724 km south of Florida, see Fig 1.1. The largest island, Grand Cayman, has a population of 32448 (CI Economics and Statistics Office, 1995) and, an area of approximately 197 km². Cayman Brac is 39 km² with a population of 1441. The smallest island, Little Cayman, is about 28 km² and has a permanent population of 33 people (CI Census 1989, 1990).

Fig 1.1 Geographic location of the Cayman Islands in the northwest Caribbean region (modification of the National Geographic Map of the Caribbean, 1975 in Ng, 1989).
George Town, the capital, is located on the west side of Grand Cayman and it is there that the major industries of tourism and finance have developed rapidly over the last twenty years. Although classified as a developing country, the Cayman Islands have a high per capita income second only to Bermuda in the Americas. Population growth, including tourism, and the resulting demand for water produced a serious deficit in the available fresh water in the Cayman Islands in the late 1970’s and 1980’s. This problem is managed by the Water Authority-Cayman (WAC) and associated desalination companies by the production of fresh water from saline groundwater.

Residential water consumption per capita is estimated at 190 l/day (van Genderen, 1991) and tourist consumption at 350 l/day (van Zanten, 1995). Major development of public water supply and sanitation infrastructures did not commence until the 1980’s. By the end of 1994, 90% of the population in the Cayman Islands had access to potable, piped water supplies.

1.2 Water Supply Development in Grand Cayman

Prior to 1988, the only area of the island served by a piped water supply was the West Bay Beach Road area which has the highest density of hotels and tourist-related businesses. This service is provided by a private company, through a franchise granted by the Cayman Islands (CI) Government in 1978. The remainder of the population relied on individual household rainfall roof catchment with storage in concrete cisterns, and or private wells for water sources. The 1989 census report (CI Census 1989, 1990) stated that 30% of the population had access or were connected to desalinated water via pipe mains (including private production plants), 51% relied on cisterns, and 18% on wells and other sources (undefined).

The average annual rainfall in the Cayman Islands during the period 1988-1993 was 1430 mm (CI Compendium of Statistics, 1994) with 1524 mm of rainfall on Grand Cayman in 1993 (CI Annual Report 1993, 1994). The dry season is from December to and including April and the wet season from May to November. It is therefore no coincidence that the tourist season is during the driest period of the year, as that is the time of the most pleasant climate. The CI Government recognised that with the rapid increases in tourist, commercial, and residential developments the demand for potable water might create a crisis if some action was not taken to satisfy that demand. Consequently, in November, 1981, the Water and Sewerage Project Office was established with technical and financial assistance from United Nations Development Programme (UNDP) and the CI Government. Through this project, three fresh groundwater lenses were identified and evaluated for further development.
The fresh water lenses located in Lower Valley and East End, Fig 1.2, were developed in 1983 and 1985, respectively. Provisions were made for monitoring (MacAree, 1991) treatment, storage and delivery of water by private trucking companies to residents who needed to supplement their private supplies.

Fig 1.2 Hydrogeological setting of the three major fresh water lenses on Grand Cayman. (Ng, 1989).

At the end of 1994 the Lower Valley wellfield abstraction and treatment facility was decommissioned in order to protect the lens from damage as the safe yield had been reached and the quality, in terms of salinity, was degrading. As part of the Authority’s piped water supply development, the East End wellfield (3000 m³/day estimated safe yield) will be further developed in 1995 to provide a small piped system for the district community.

The Water and Sewerage Project Office became the Cayman Islands Water Authority in May, 1983, with the passing of the Water Authority law. This law is "...to provide for the establishment of a Water Authority to provide for the membership, management, power and duties thereof, in relation to water supply and sewerage in the Islands, to
provide for the licensing of existing and new water abstractions and for the control thereof, to provide for the control of water pollution and water supplies, to control sewage disposal, for the licensing of well diggers and for matters connected therewith and incidental thereto." (WA Law, 1982). In 1990, the Water Authority-Cayman became a full-fledged "Statutory Body" with the power to enforce regulations provided by law regarding groundwater management; wastewater collection, treatment and disposal; and public water supplies.

In 1985, the Water Authority and the Environmental Health Department carried out a study of 200 household wells being used in the capital, George Town. The results indicated that 65% of the wells were faecally contaminated and 47% had chloride levels of > 600 mg/l (Beswick, 1985). Additionally, the incidence of gastroenteritis occurring in children (Grand Cayman) under the age of 5 year was reported and is shown in Table 1.1 below.

Table 1.1 Reported cases of gastroenteritis among children (aged under 5 years, Grand Cayman), (adapted from Beswick, 1985)

<table>
<thead>
<tr>
<th>YEAR</th>
<th>NUMBER OF CASES</th>
<th>ESTIMATED POPULATION*</th>
</tr>
</thead>
<tbody>
<tr>
<td>1980</td>
<td>91</td>
<td>17760</td>
</tr>
<tr>
<td>1981</td>
<td>45</td>
<td>18200</td>
</tr>
<tr>
<td>1982</td>
<td>46</td>
<td>18700</td>
</tr>
<tr>
<td>1983</td>
<td>71</td>
<td>19300</td>
</tr>
<tr>
<td>1984</td>
<td>406</td>
<td>20200</td>
</tr>
<tr>
<td>Jan - Nov 85</td>
<td>318</td>
<td>21000</td>
</tr>
</tbody>
</table>

(* data from CI Compendium of Statistics, 1994)

For the 1985 data reported in Table 1.1, about 50% of the patients were recorded as living in George Town. The suddenly increasing trend from the 46 cases in 1982 to 318 for 9 months in 1984, although not directly correlated with water quality, was cause for considerable concern. The health risk implications of a population using faecally contaminated water for potable use have been well documented (Baltazar, et al, 1988; and Daniels, et al, 1990).

From a survey conducted by the Water Authority in the George Town community, 80% expressed positive interest in a piped water supply. The George Town Water Supply Project was developed as a result of the survey and also as a result of the study conducted on the potable water quality of wells being used by people in the community.
In 1987, construction of reservoir, pumping station and laying of pipes began, with the first delivery of piped water in February, 1988. Because of the tremendous demand, extensions resulted in over 4500 connections (at end of 1994) serving more than 17000 people in George Town and the eastern district of Bodden Town. The water supply project was completed in May, 1994 with more than 180 km of polyvinyl chloride (PVC) and polyethylene (PE) pipes in the ground.

Water in the islands is costly, the residential rate for the first 12 m$^3$ is US$5.00/m$^3$ for piped water (1995 Water Authority rate). This reflects the expensive process of reverse osmosis and steam distillation used to produce potable water from saline groundwater. This is necessary as the islands have no surface water and the remote location of the fresh groundwater lenses in relation to George Town, make it uneconomical to pump water. The average daily water demand for the distribution system at the end of 1994 was approximately 3000 m$^3$/day (van Genderen, 1995). Cayman Water Company, the company holding the private franchise for water supply in the West Bay Beach Road area extended their services to the district of West Bay in 1992. The average daily water demand in the West Bay Beach Road area at the end of 1994 was 1935 m$^3$/day while in the residential district of West Bay it is 435 m$^3$/day (McCoy, 1995).

1.3 Water Supply in the Sister Islands
In order to assist the island of Cayman Brac with its development, the Government financed the Water Authority-Cayman to provide a piped water supply serving 51 customers over a pipe length of 4 km. The distribution system, commissioned in June 1991, distributes water produced by a 230 m$^3$/day capacity reverse osmosis plant at a cost of US$6.30/m$^3$ (1994 Water Authority rate). Because this island has limited annual rainfall (mean of 1054 mm/year) and only one small freshwater lens which is largely inaccessible to a majority of the population, it was expected that provision of piped water supply infrastructure would be a catalyst to encourage development.

1.4 Sanitation Development in the Cayman Islands
In the Cayman Islands on-site sewage disposal has been the norm for as long as the Islands have been inhabited. In the middle of this century, inside plumbing and the flush toilet became more popular than the traditional outhouse (pit latrine). There are a few remaining pit latrines in use, however, the majority of the population uses septic tanks. The percentage of residents using different types of sewage disposal is shown in Table 1.2. Prior to 1981, wastewater from the bathroom was removed and stored in cesspools, while wastewater from washing and the kitchen was led to a soak-away.
This situation is thought to be one of the contributing factors to the poor quality of groundwater now found in George Town.

In Grand Cayman, the groundwater table ranges from 0.3-10 m below the ground surface, thus, the risk of contamination from pit latrines, cesspools, and soak-aways is high. In addition, soak-aways, if not properly maintained tend to provide a hospitable breeding ground for mosquitoes and other nuisance insects.

<table>
<thead>
<tr>
<th>SEWAGE TREATMENT</th>
<th>PERCENTAGE OF RESIDENT POPULATION</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Grand Cayman</td>
</tr>
<tr>
<td>Waste stabilisation ponds/sewers</td>
<td>7%</td>
</tr>
<tr>
<td>Package plants</td>
<td>0.5%</td>
</tr>
<tr>
<td>Septic tanks</td>
<td>91% combined with septic tanks</td>
</tr>
<tr>
<td>Cesspools</td>
<td>22%</td>
</tr>
<tr>
<td>Pit latrines</td>
<td>0.7%</td>
</tr>
<tr>
<td>Other</td>
<td>0.8%</td>
</tr>
</tbody>
</table>

The issue of groundwater contamination was addressed by the Water and Sewerage Project Office in conjunction with the Central Planning Authority and Environmental Health Department (EHD) in 1981. Standardisation of on-site domestic sewage disposal systems was agreed between the three departments. This resulted in septic tanks being built according to agreed specifications and the final effluent disposed of in deepwells cased to a minimum depth of 17 m. All on-site disposal must be situated more than 17 m away from cisterns and wells. Deepwells were chosen as the most acceptable method in which to dispose of septic tank effluent because:

- the topsoil cover is thin (<0.1m in many places)
- the calcareous rock is usually fissured and cracked
- the depth chosen is into the brackish/seawater aquifer
- little maintenance is required, other than emptying tank of sludge

On account of the rapid development in the Islands and responsibility as charged by the WA Law, the Authority continues to be actively involved in sewage treatment and disposal at the design stage for developments larger than a residential duplex. The
sewage treatment method must meet the approval of the Development Control Unit of the Authority prior to Central Planning Authority approving a project.

For all developments, whether residential or commercial, a 'Certificate of Occupancy' is issued by the Central Planning Authority before the premises are occupied. This certificate is issued when all planning requirements have been met and premises have been inspected by the Planning Department. If these requirements are met then it is presumed that the development will provide a safe and healthy environment with regard to water supply, waste disposal, electrical, plumbing, and structural elements.

It is recognised by the Authority that disposal within the saline aquifer may influence the coastal waters. Monitoring of faecal indicator bacteria in the George Town Hog Sty Bay Harbour, is carried out in conjunction with the Protection and Conservation Unit (PCU) of the Department of the Environment (DoE). Results of this programme, ongoing since 1991, have not indicated land-based wastewater pollution (WA Annual Report 1993, 1994) in this area and average contamination levels remain at <1 faecal coliform cfu/100 ml (WA Annual Report 1994, 1995).

1.5 Sewage Treatment in the Sister Islands
The use of septic tanks was extended to the sister islands with the same requirements for effluent disposal as in Grand Cayman. Cayman Brac has a dry, rocky terrain, and a very small freshwater lens which is used for watering cattle, albeit major development of this lens is not envisaged in the near future. Little Cayman is covered by extensive areas of swamp with no fresh groundwater yet discovered. Unless these islands develop rapidly, there will be little demand to introduce a public wastewater collection and treatment system in the near future.

1.6 Development of Sewerage System and Sewage Treatment in Grand Cayman
As the majority of tourism oriented industry is centred on the West Bay Beach area, major financial investments such as hotels, condominiums and other tourism-supporting businesses have concentrated there. The rate of growth and development in the area was more rapid than development of supporting sanitation infrastructure. This resulted in establishments in the area utilising package plants or septic tanks to treat wastewater. Numerous problems were experienced with package plants. They were often poorly operated (Fig 1.3) and not maintained causing emission of offensive odours. Occasionally the effluents overflowed onto premises creating an unpleasant and unsanitary sight.
Fig 1.3 Poorly operated sewage treatment package plant at a West Bay Beach condominium complex showing excess solids accumulation at the inlet.

The final effluents of these plants were disposed of in deepwells situated on the establishments' premises. The average volume of wastewater disposed of in the deepwells was estimated to be 1200 m$^3$/day in 1983 (CDB, 1991). Because in this area the saline groundwater table is high in several places (1 m - in relation to surface ground...
level) and close to the sea there was concern that the marine environment was being polluted by the disposal method practiced.

Beautiful beaches and aesthetic marine environment form the main attractions of the West Bay Beach area. Thus it was out of concern for protecting this segment of the islands' economy that the risk of wastewater pollution to the environment was evaluated. Risk assessment surveys carried out through literature search and marine water bacteriological analyses by the WAC in 1985 indicated that the environmental and health risks were unacceptably high. In fact, high enough to warrant intervention in order to protect the fragile tourist dollar. With the goal of protecting the environment upon which the economy largely depends, the Government made the decision to provide sewerage and treatment for that area of Grand Cayman in 1985.

It was expected that provision of sewage collection and treatment infrastructure would benefit the residents of the area, not only by reducing human and environmental health risks, but would also provide savings by removing the necessity of constructing and maintaining on-site treatment systems (CDB, 1991).

Waste stabilisation ponds were chosen as the method by which to treat sewage collected from the West Bay Beach area. This was done after a least cost analysis was carried out by staff of the funding institution - Caribbean Development Bank (CDB) and WAC engineers on the two feasible treatment plant options; an oxidation ditch system and a waste stabilisation pond system (CDB, 1991).

In choosing the waste stabilisation treatment pond system, a number of factors were taken into account by the Water Authority:

a) Simplicity of operation and maintenance as compared to conventional sewage treatment methods.

b) Low energy requirements (without aeration).

c) Design equations predicted adequate pathogen removal by the pond design selected.

d) The possibility of earning revenue through irrigation reuse of final effluent on a nearby golf course.
e) Reducing the public health and environmental risks by providing disposal and treatment of septage in a contained area instead of drying on land.

f) Waste stabilisation ponds have a large buffering capacity and are therefore able to absorb shock loads from septage.

g) Optimal performance of system was assured because of the tropical climatic conditions on the island.

The construction of the sewerage and treatment system began in 1986 and was completed in 1988 at a cost of US$11.9 million (CDB, 1991). The system was commissioned in February, 1988. It consists of 15 km of 100-450 mm diameter gravity sewers (vitrified clay and PVC), about 450 manholes, 17 sewage lift stations and a major pumping station which pumps sewage 2 km to the sewage treatment works via a 400 mm ductile iron pressure main (McTaggart, 1995). To date (1994), there are 269 connections of which 154 are residential, 108 commercial, and the remaining 7 customers are septic tank emptiers (truck pumping).

In order to ensure that the effluent met the criteria set for irrigation and that the sewage collection and treatment plant were performing properly, a routine monitoring programme was set up, commencing with the commissioning of the system in 1988. This routine monitoring programme evolved to suit the needs of the WAC as problems with saline groundwater intrusion into the sewers became evident and became a major factor affecting the performance of the waste stabilisation ponds.

The drainage area and location of treatment works are shown in Fig 1.4.
Fig 1.4 Map showing location of West Bay Beach sewerage and treatment works (WAC, 1985).
1.7 Justification of Study
The potential for reuse was one of the considerations in choosing waste stabilisation ponds. This was expected to be beneficial from two points of view; water resource reclamation, and as a revenue-earner because the projected income was estimated at US$12,000 per month in 1988. The management of a nearby golf course was willing to purchase the treated pond effluent, providing that the salinity was <2500 μS/cm and the faecal coliforms were ≤1000 cfu/100 ml.

1.7.1 Objectives of Study
The original objective of the monitoring programme was to ensure that the effluent quality was such that the criteria for reuse were met.

A. Practical Objectives
Establish a monitoring programme in order to:

1. Permit regular assurance that the effluent quality is complying with the Authority’s discharge and reuse standards.

2. Ensure that the effluent is not harmful with respect to environmental health and the receiving groundwater.

3. Provide insight into the probable cause if the pond system suddenly fails or the effluent quality deteriorates.

4. Generate data to provide information on the loading of the waste stabilisation pond system, whether underloaded or overloaded. This information is useful to determine whether the loading may be safely increased as the community expands or whether further ponds are needed.

5. Establish performance data including sludge accumulation rates and faecal indicator and biochemical removal constants, under local conditions. These could be used in design considerations for expansions to the existing system in the future and that may assist in improving the design of future ponds in the Cayman Islands by taking into account the local conditions. The adaptation of waste stabilisation pond design to the local climate is particularly important because they utilise processes which are dependent on natural physical, chemical and biological mechanisms for wastewater quality improvement/stabilisation.
B. **Academic Objectives:**

1. Increase scientific understanding of the mechanisms operating in lagoons and the factors influencing the performance in terms of biochemical oxygen demand and removal and pathogen indicator removals operating under the Cayman conditions.

2. During the last 50 years design equations currently in use have been developed based mainly on the empirical approach. In order to improve efficiency and more accurately predict effluent quality it is essential that fundamental scientific knowledge of mechanisms controlling and contributing to the behaviour of waste stabilisation ecosystems be further expanded and refined.
CHAPTER 2

2.0 GENERAL LITERATURE REVIEW

2.1. Introduction

This thesis is primarily concerned with increasing the scientific understanding of one of the most important sanitary barriers to human disease; wastewater treatment by lagooning.

The great majority (80%) of human infectious disease in developing countries is associated with poor sanitation (Anon., 1980). This realisation led to a concerted international effort to improve water supply and sanitation during the decade 1980-1990 through the "International Drinking Water Supply and Sanitation Decade (IDWSSD)".

A basic principle of sanitation, as applied to human waste, is that containment of excreta for sufficient time will render it harmless because pathogens and parasites die out. This principle finds a major application in rural areas worldwide in the form of pit latrines. However, in a great majority of towns and cities waterborne sewerage is used to carry excreta away from the site of defaecation and introduces a greater order of complexity into excreta disposal practices. The use of water to transport human wastes has the benefit of convenience but immediately increases the problems of containment and treatment.

When the human population was small and dispersed, defaecation "in-the-bush" presented little risk of reinfection, but the population growth that has occurred during the 19th and 20th Century has resulted in dense urban communities. These crowded communities demand sophisticated, elaborate and well-organised waste collection and treatment in order to avoid massive epidemics of disease.

The World Health Organisation (1987a) estimated that 85% of rural people and 14-15% of people in urban areas still do not have access to adequate sanitation. By contrast, the efforts to provide water supplies were claimed to be successful for 86% of urban people and 44% of rural people. During the IDWSSD the increased number of water and sanitation facilities did not keep pace with population growth. As a consequence, there are more people unserved at the end of the decade than in 1980, although in percentage terms, percentage coverage has increased. Furthermore, sanitary disposal of human
waste has lagged behind water supply, which implies that more human excreta is contaminating the environment in the 1990s than in the 1980s and more people in the developing world are without adequate sanitation (WHO, 1988).

2.2 Development of Urban Sanitation

The provision of appropriate sanitation and safe water supply for a community, be it large or small, has been shown to have major health benefits such as reductions in diarrhoeal morbidity and mortality rates (Esrey et al, 1985). Investments undertaken to improve water quality and provide for excreta disposal are undeniably effective in reducing death and sickness from diarrhoea as demonstrated in Table 2.1.

Table 2.1 Percentage reductions in diarrhoeal morbidity rates attributed to water supply or excreta disposal improvements (adapted from Esrey et al, 1985).

<table>
<thead>
<tr>
<th>Type of intervention</th>
<th>Number of studies</th>
<th>Median</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>All interventions</td>
<td>53</td>
<td>22</td>
<td>0-100</td>
</tr>
<tr>
<td>Improvements in water quality</td>
<td>9</td>
<td>16</td>
<td>0-90</td>
</tr>
<tr>
<td>Improvements in water availability</td>
<td>17</td>
<td>25</td>
<td>0-100</td>
</tr>
<tr>
<td>Improvements in water quality and availability</td>
<td>8</td>
<td>37</td>
<td>0-82</td>
</tr>
<tr>
<td>Improvements in excreta disposal</td>
<td>10</td>
<td>22</td>
<td>0-48</td>
</tr>
</tbody>
</table>

*There are 53 results in total but only 44 attributed to specific interventions. The remaining 9 results are for other interventions or combinations of interventions.*

The reduction of diarrhoea rates in children of poor urban and rural communities has been well documented after improvements in the water supply and sanitation occurred (Baltazar et al, 1988; and Daniels et al, 1990). Not only does the community's health improve, but time that was otherwise utilised for water collection or caring for the sick, becomes available for leisure, home improvements, and work. This benefits communities by improving the quality of life for its citizens.

However, providing communities with adequate supplies of water and appropriate sanitation is not a simple, straightforward task. One of the major inhibiting factors is the high per capita cost especially when waterborne sewerage is being considered. The
method of sanitation used is often dependent on the quantity of water available. In dense urban areas, normally water is available on tap, therefore, the flush toilet is the most convenient and is frequently used. By effectively diluting waste >100-fold, water makes the waterborne system the most demanding of water resources in a country.

Wastewater is typically composed of 99.9% water with 0.1% suspended settleable and colloidal, and dissolved solids - organic and inorganic compounds (IRCWD, 1988; Gray, 1989). It is estimated that 30-40% of the domestic water consumption is used for toilet flushing (Mara, 1982). However, in developing countries, flush toilets are very often beyond the affordability of low income people due to the cost of installation and upkeep. Not only is the toilet and the accompanying plumbing necessary but the availability of a constant and abundant supply of water is vital. Consequently, lower income communities and/or rural societies often utilise a variety of on-site excreta disposal facilities such as pit latrines and pour-flush toilets. Thus water supply and socio-economic level will strongly influence the type of wastewater removal and treatment used within communities.

For treating wastewater there are two fundamental objectives:

1) To protect the human population from health hazards associated with wastewater, e.g. by removal of pathogens and parasites.

2) To purify the effluent to a level that it can be returned to the natural environment without damaging it, or in a form that the water can be reused.

The first objective seeks to protect the public's health, while the second seeks to limit or avoid pollution of the environment.

The aims of sewage treatment are:

1) to separate particulate matter, including pathogens and parasites from the transporting liquid phase.

2) to convert dissolved substances into biomass which can then be readily separated from the liquid phase.

3) to safely dispose of sludge produced in 1) and 2) above.
In order to achieve these aims, treatment of wastewater in sewered areas generally fall into two major categories: "conventional" and "unconventional." It will be demonstrated in this review that "conventional" treatment is relatively ineffective in removing pathogens and parasites.

To effect the treatment of wastewater, the 5 stages of treatment processes can be classified into the following units:

I. Preliminary Treatment - removal of grit and gross solids
II. Primary Treatment - physical sedimentation
III. Secondary Treatment - biological oxidation and synthesis of biomass and settlement
IV. Tertiary Treatment - polishing to improve effluent quality to high standards
V. Sludge Treatment - dewatering, stabilisation and disposal

The final disposal of the effluent should determine how many unit treatment processes the wastewater will go through. However, it may be noted that most coastal towns (even today) in Europe do not treat wastewater before disposing of it by sea outfalls. They depend on dilution at sea which is becoming increasingly unacceptable (EEC, 1988).

2.3 Development of Conventional Treatment

Conventional treatment in Europe and North America is generally considered to include primary sedimentation and treatment by means of either percolating filters or activated sludge. The origins of conventional treatment came about as a result of the industrial revolution in the United Kingdom (UK) during the 19th Century. As population densities increased, the need for drainage in order to prevent stormwater flooding populated areas became obvious to city authorities. The provision of sewers designed to carry only storm waters resulted. Nonetheless, the disposal of human excreta and other wastes into these drainage systems was widely practised.

In 1849, the UK authorities took the decision to legalise faeces disposal into the sewers. This action, in conjunction with the increasing population densities, produced gross contamination of drinking water sources and gave rise to the notorious cholera epidemics in the last quarter of the 19th century because sewage received no form of treatment. The middle of the 19th century saw the development of the first large, modern sewers in Hamburg, Germany (1844-48) and London (1854-65), (Coffey and
Reid, 1978). Epidemics arose mainly due to the sewers discharging into rivers which were utilised downstream for drinking water purposes. The Public Health Act in 1875, defined the responsibilities of local authorities in the UK for disposal and treatment of sewage.

By 1898, the UK Royal Commission on Sewage Disposal was set up and eventually recommended that sewage be treated to a standard of 20 mg/l BOD (biochemical oxygen demand) and 30 mg/l SS (suspended solids). In 1912, the 8th report of the Commissioners appointed to administer the Royal Commission on Sewage Disposal (1912) was accepted governing the discharge of treated sewage effluents to the 20/30 limit. The effluent discharge regulations were concerned with discharges to rivers. In addition to the 20/30 effluent quality, the receiving river should provide an 8-fold dilution of the treated sewage effluent (Klein, 1959). Then, as now, there was no microbiological standard for conventional wastewater effluents. The science of microbiology was in its infancy in the 1890's when the bacteriological monitoring of drinking water quality was introduced using the gelatine plate count, for example, by Frankland (1886) in London.

The early sewage treatment methods included broad treatment, contact beds and percolating filters in the 1880's (Coffey and Reid, 1978). In 1882, William Joseph Dibdin, a chemist in the UK, developed the idea of a filtering medium to clean sewage. This led to the well known percolating filter that is used extensively in industrialised countries. Then, in 1913, the activated sludge process was invented by Arden & Lockett at the Davyhulme Treatment Works in Manchester, England (Gray, 1989). The idea of activated sludge began in 1912, with Dr. Gilbert Fowler and his assistants at the Manchester Rivers Department (Coffey and Reid, 1978); it remains the most widely used, and largest controlled microbiological wastewater treatment process today. The percolating filter and activated sludge processes with their various adaptations have since become known as "conventional sewage treatment methods" to the industrialised world.

2.3.1 Disadvantages of Conventional Sewage Treatment

The term conventional treatment at present refers to relatively high-rate primary sedimentation, secondary biological treatment by activated sludge or percolating filters followed by secondary sedimentation. Generally, the resulting sludge accumulation is separated and treated by anaerobic digestion and drying beds. Unconventional treatment follows the same principles of primary sedimentation and secondary biological treatment in lagoons but the processes are less controlled and understood.
Additionally, sedimented sludges are not routinely separated from the supernatant liquid.

Mara (1982) identifies 3 major disadvantages in using 'conventional' treatment in developing countries:

1) Relatively poor pathogen removal efficiencies: for example; poor removal of salmonellas as observed by Yaziz and Lloyd (1979, 1982);

2) High capital and running costs (usually with the need to import all or much of the mechanical equipment, with a consequent foreign exchange cost); and

3) The requirement for a high level of operating and maintenance skills.

Although conventional works are capable of meeting the Royal Commission discharge standards, they are unable to remove pathogens and indicator bacteria to the degree that would permit reuse of wastewater. The Engleberg Report in 1985 (IRCWD, 1985) recommended that treated domestic wastewater effluents reused for unrestricted irrigation should contain ≤1 viable nematode egg/l and ≤1000 faecal coliforms cfu/100 ml, using the geometric means. Unless supplemented by disinfection, conventional processes are unable to produce effluents that meet the Engleberg criteria for unrestricted irrigation (IRCWD, 1988). The so called 'unconventional' methods are able to reduce pathogens to the levels specified in the Engleberg Report and therefore present much lower levels of risk to public health.

2.4 Lagooning - Unconventional Treatment Method

The process of sewage treatment by waste stabilisation ponds is normally considered as unconventional by the industrialised world. Ironically, stabilisation lagoons or oxidation ponds, as they are frequently referred to, have been used in varying forms for hundreds of years by communities in Asia (Mortimer, 1954). Allum and Carl (1970) point to historical data showing that almost a thousand years ago, the Chinese were using rudimentary waste stabilisation ponds. The primary purpose of these ponds was not the treatment of human waste but the cultivation of fish! In the early part of the 20th century, Germany was utilising sewage ponds for fish cultivation (Mortimer, 1954). Many parts of Asia still utilise raw sewage-fed fish ponds in aquaculture!

The initial utilisation of lagoons for the treatment of sewage in the United States (USA) apparently resulted from an emergency discharge into a basin dug in an old creek bed at
Santa Rosa, California in 1924 (Caldwell, 1946; and Neel and Hopkins, 1956). Giesecke and Zeller (1936), described the treatment of sewage in a holding lake with positive results. The first pond considered to be built on sound engineering principles in the USA was constructed in 1948 (Porges and Mackenthru, 1963). In the USA, there are almost 7000 waste stabilisation ponds being used for wastewater treatment (USA EPA, 1983), while in Canada, over 1000 plants are in operation (Gray, 1989). Although, often referred to as 'unconventional', throughout the world waste stabilisation ponds have gained recognition and acceptance as an efficient, low-cost method of domestic and industrial wastewater treatment. In Melbourne, Australia, sewage treatment through the use of lagoons dates back to 1936 (Hussainy, 1979). Mitchell (1980a) reported that maturation ponds were used by 300 towns throughout Australia. According to Gloyna (1971), by 1970 ponds were used in 39 countries ranging from the polar regions to the equator serving populations ranging from <1000 to >100000.

Prior to the 1950's the use of lagoons was not encouraged in the USA (Heuvelen, 1970) however this changed in the early part of that decade when the Missouri River Basin Engineering Health Council was established. This Council established and published design criteria, construction and operation practices for the Missouri river basin area. In 1974, Martinez reported that there were 260 waste stabilisation pond systems operating in Cuba (Martinez, 1974). More recently, it is reported that in the early 1980's there were 561 lagoon installations in the Caribbean and Latin America alone (Saenz, 1985). More than 3500 treatment lagoons are used in Germany and France (Vuillot and Boutin, 1987). Marecos do Monte (1992) reported that waste stabilisation ponds can be found in 19 European countries: Sweden; Finland; Denmark; Holland; Belgium; Germany; Switzerland; Poland; Hungary; Czechoslovakia; Romania; USSR; Turkey; Cyprus; Italy; France; Spain; Portugal and the UK.

2.5 Waste Stabilisation Ponds (WSP)

Waste stabilisation ponds or lagoons are defined as shallow dyked structures designed specifically to treat wastewater by "self-purification" utilising natural biological, chemical, and physical processes. The behaviour and treatment efficiency of a pond system depends on natural factors such as solar radiation, wind action, temperature, rainfall and evaporation.

Waste stabilisation ponds have been defined by Hawkes (1983) as:
"any natural, or more commonly, artificial lentic (i.e. standing) body of water in which organic wastes (either crude sewage, settled, organic and oxidisable industrial effluents or oxidised sewage effluents) are treated by natural biological, biochemical and physical processes commonly referred to as 'self-purification' or stabilisation."

These structures although, mechanically simple compared to conventional wastewater treatment methods, are very complex biotic systems. Once operational, there is little that an operator can do to control the processes' performance, since their physical dimensions and climatic conditions exert the greatest influence. It is thus, on these two factors that the sanitary engineer bases the pond system design. There are a variety of design equations in use presently and these will be discussed further on in this chapter. Ponds are used as primary, secondary and/or tertiary treatment systems of both domestic and industrial wastewater.

In terms of economic savings waste stabilisation ponds are in the forefront of all treatment methods. They are unrivalled in the quality of the final effluent as their self-regulating potential is high. The appropriately designed pond system will provide a steady, effective, low-cost and technologically simple means for treating domestic and some industrial wastewaters (Arthur, 1983). The costs of treating sewage in waste stabilisation ponds compared to that of conventional treatment plants has been analysed by several authors. The cost of waste stabilisation pond treatment is generally cheaper when the cost of the land is not prohibitive (van Zanten, 1991; Marecos do Monte, 1992; and Xian-wen, in press).

The design of oxidation pond systems, as with most wastewater treatment processes, is directly related to the environment receiving the final effluent. For instance, an effluent that will be disposed of by river discharge will have different quality requirements to that which will be reused for irrigation of food crops. Faecal coliform indicator bacteria removal in conventional treatment plants are normally in the range 90-98% (Carrington, 1980) while removal in waste stabilisation pond systems are often 99.99% (Mara, 1976). A comparison of the survival of potential pathogens in effluents from conventional treatment works and a typical waste stabilisation pond system is reported in Table 2.2:
Table 2.2 Summary of range of concentrations of potential pathogens surviving in effluents after various sewage treatment processes (adapted from Lloyd, 1982).

<table>
<thead>
<tr>
<th>Treatment Process</th>
<th>E. coli/litre</th>
<th>Salmonella/litre</th>
<th>Enteric viruses/litre</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Primary sedimentation</td>
<td>$10^0-10^7$</td>
<td>$10^1-10^3$</td>
<td>$10^0-10^5$</td>
</tr>
<tr>
<td>2. Percolating filter with 1° &amp; 2° sedimentation</td>
<td>$10^0-10^7$</td>
<td>$10^2-10^3$</td>
<td>$10^2-10^4$</td>
</tr>
<tr>
<td>3. Activated sludge with 1° &amp; 2° sedimentation</td>
<td>$10^4-10^5$</td>
<td>$0-10^3$</td>
<td>$10-10^4$</td>
</tr>
<tr>
<td>4. Raw sewage stabilisation ponds with 3 cells and 15-25 day retention</td>
<td>$10-10^4$</td>
<td>$0-10$</td>
<td>$0-10$</td>
</tr>
</tbody>
</table>

Advantages of using WSP over conventional treatment methods are summarised by Mara (1976) as follows:

1. Ability to achieve any degree of purification with the minimum maintenance by unskilled operators.
2. Removal of pathogens is considerably greater than that achieved by conventional sewage treatment methods.
3. Tolerance of both hydraulic and organic shock loads.
4. Ability to effectively treat a wide variety of industrial and agricultural wastes.
5. May be designed so that the degree of treatment is easily altered.
6. Method of construction is such that the land can easily be reclaimed in the future.
7. Algae produced in the ponds are a source of high protein food which can be easily used for pisciculture.

Some of the main disadvantages of stabilisation ponds summarised by Shelef and Kanarek (in press) are:
1. High land costs and large land area required.

2. Effluent high in algal cells - controversial discharge to receiving waters.

3. Performance is dependent to a large extent on climatic conditions such as temperature, solar radiation, wind velocity, etc.

4. Overloading or abrupt climatic changes can cause odour nuisances and deterioration of effluent quality.

Classification of ponds is conveniently dependent on the dominant type of biological activity taking place in that particular pond. Greater reduction in pathogens and BOD are possible when ponds are operated in series. The first pond in series (either anaerobic or facultative) will have a higher organic loading and sludge accumulation. There are 3 main pond types (anaerobic, facultative and maturation), plus speciality ponds such as advanced integrated pond systems (AIPS), mechanically aerated, macrophyte, high-rate algal and pisciculture ponds (Fig 2.1).

As WSPs are considered to be very hardy and able to withstand both hydraulic and organic shock loads, they are able to treat a wide variety of industrial wastewaters with up to 60 mg/l heavy metals (Zickefosse and Hayes, 1977) which precipitate out as sulphides in anaerobic ponds or as hydroxides in aerobic ponds (Alabaster et al, 1991). Treatment of industrial wastes containing compounds such as phenol-based
hydrocarbons is not recommended because of algal inhibition (Arthur, 1983). Strong organic wastewaters from agro-industries such as abattoirs, piggeries, food canneries and dairies are easily treated in WSPs (Mara et al, 1992a; and Oliveira et al, in press).

The main pond types and the mechanisms by which they work and are designed are discussed further.

2.5.1 Anaerobic Ponds

These ponds may be best described as pretreatment ponds which function in a similar way to open septic tanks. They are characterised by retention times of 1-5 days and depths of 2-4 m (Feachem et al, 1983; and Horan, 1990). Parker et al (1950) were among the first to describe these ponds as 'anaerobic'. The classification 'anaerobic' describes one of the conditions that is characteristic of these types of ponds, although sedimentation is the certainly the most important activity. These ponds reduce, by digestion, the biodegradable organic material found in strong domestic and some industrial wastewaters.

There are few guidelines in literature for the design of anaerobic ponds. However, it is generally considered acceptable to design anaerobic ponds based on the assumption of complete mixing and the volumetric loading of BOD which is given by:

\[ \lambda_v = \frac{L_t Q}{V_a} \]  

\( \lambda_v \) = BOD loading g/m³ d

Meiring et al (1968) and Arthur (1983) recommend that the volumetric BOD loading should be between 100-400 g/m³ d, however Mara et al (1992a) recommends an upper limit of 300 g/m³ d in order to provide a safety margin with respect to odour. Other pond designers and researchers (Yanez et al, 1980; and Eckenfelder and Englande, 1970) have suggested that the design be based on surface loading rate (BOD kg/ha d).
Removal of BOD from the liquid phase is accomplished primarily by the action of the pond as a settlement tank. Retention time and temperature are major factors that control BOD removal in anaerobic ponds. In temperate climates with temperatures below 15°C, BOD removal is due principally to settlement as little anaerobic breakdown of organic matter occurs. However, in the summer when temperatures rise above 15°C, the accumulated sludge is digested. Parker et al (1950) reported 47.1-82.2% BOD removal in anaerobic lagoons in Australia. In California anaerobic ponds, 50-80% BOD removal was reported by Roesler and Preul (1970). Gomes de Sousa (1987) described 45% BOD reduction in experimental anaerobic lagoons in Portugal.

It could be argued that excessive retention time (>24 hours) in temperate climates is counterproductive in anaerobic ponds because the main mechanism for BOD removal is sedimentation and this occurs in the first few hours as in conventional treatment. The BOD removals reported for anaerobic ponds are only slightly better than the reductions reported for the primary treatment phase in conventional processes which are achieved in 4-8 hours. This is at odds with earlier claims of 'super' efficiency of WSPs because lagoons are expected to have the capacity to sediment continuously over many years without a daily desludge cycle, whereas primary sedimentation in conventional treatment is maintained intensively by 'fill and draw'. The process of sedimentation in anaerobic lagoons is summarised in WHO (1987b), (Fig 2.2):

![Stabilisation process in an anaerobic pond](image-url)

Fig 2.2 The stabilisation process in an anaerobic pond (WHO, 1987b).
Gray (1989) reported a reduction in BOD of between 25-40% and 50-70% of suspended solids during the primary treatment phase in conventional plants. For anaerobic ponds operating in climates with temperatures above 20°C and with 300 g BOD/m³ d design loading, BOD removals are expected to be 60% according to Mara and Marecos do Monte (1990), although Ellis (1983) refers to 25-30%. Hammer (1977) reported that with a 320 g BOD/m³ d loading, minimum temperatures of 25°C, and retention time of 4 days that 75% removal of BOD is possible.

Experimental studies have been carried out on deep (3.40 m) pilot-scale anaerobic ponds at the Experimental Station for the Biological Treatment of Sewage (EXTRABES), Brazil, by Oragui et al (1987). These studies showed that with a loading of 215 g BOD/m³ d and a retention time of 1 day, 83% of the BOD was removed. Results such as these have implications for the design of ponds in the future which are smaller, therefore requiring less land and consequently, opening up the feasibility of utilising lagoons in dense, urban areas or areas with a limited amount of flat land available. An example of the latter is St Lucia where ‘Advanced’ lagoon systems of the type promoted by Oswald (1991) and Green et al (in press) have been integrated into the design, with considerable saving on land requirements. These types of lagoons, known as AIPS are generally described as an anaerobic pond within a facultative pond. They have been promoted in California for over 20 years by researchers at the University of California, Berkeley (Green et al, in press).

According to Silva (1982) pathogen removal in anaerobic ponds is due to sedimentation and appears to be largely independent of loading and retention time. Settlement in anaerobic ponds of suspended solids will include helminths ova, parasites and solids associated bacteria. These solids settle to the bottom of the pond and then undergo anaerobic digestion, concentration and some mineralisation. This process results in the emission of methane, hydrogen sulphide, and carbon dioxide gases into the pond liquor and dispersion into the atmosphere follows through the liquid surface.

In anaerobic ponds saprophytic bacteria decompose organic matter forming malodorous volatile organic acids which are in turn metabolised by methanogenic bacteria. Generally it is the concern of malodorous gaseous emissions that discourages design engineers from including anaerobic units in pond treatment systems. Odour formation should be minimal when anaerobiosis is complete because, to obtain energy for growth, methanogenic bacteria pair the oxidation of volatile acids to the reduction of carbon dioxide resulting in the formation of methane gas (Horan, 1990) which is odourless.
Sulphate concentrations in the sewage influent are of particular importance in the control of odour due to the anaerobic reduction of sulphate occurring in anaerobic ponds. This reaction produces hydrogen sulphide (H₂S) which is usually the cause of the foul smell associated with anaerobic ponds. Unfortunately, odour nuisance can be experienced if the wastewater being treated is sulphate-rich (Mara, 1976) or if the rate of volatile short chain organic acid formation exceeds the ability of the methane fermenting population to metabolise it (that is, organic loading too high) (Parker, 1979).

Methanogenesis and sulphate reduction are alternate degradation reactions competing for common substrates however sulphate reduction is favoured if sulphate is present (Widdel, 1988). Sulphate concentrations of 500 mg/l and above are considered by Meiring et al (1968), Gloyna (1971) and Mara et al (1992a) to be the maximum that can be tolerated in a pond system. The issue of sulphate loading and the resulting sulphur transformations have a major significance on the study presented in this thesis and are discussed in Chapter 8.

The rate of sludge build up in anaerobic ponds is reported to be slow, 40 l per person per year (WHO, 1987b). Mara (1976) reports that sludge accumulation is approximately 0.03-0.04 m³/ha d, and desludging is required when a pond is half full. After 2 years of operation, Parker et al (1950) reported 40 cm of sludge accumulated in an anaerobic pond in Australia. They also observed that anaerobic ponds with sludge were more efficient in BOD removal than those without sludge accumulation. However as the sludge level built up, BOD removal improved. This was attributed to the development of maturation of microbial digestion in the sludge layer as organic matter is converted to carbon dioxide under anaerobic conditions. At temperatures above 15°C anaerobic digestion of the solids occur and desludging is not necessary for 3-5 years (Mara, 1976; and Pacey, 1978).

2.5.2 Facultative Ponds
Facultative ponds are shallower than anaerobic ponds and possess a much larger surface area. Retention times of 10-14 days and depths of 1.5-2.0 m are the normal characteristics of these types of ponds. Facultative ponds may be designed to receive raw sewage, septage, night-soil (primary facultative ponds) or the effluent from an anaerobic treatment pond or some other form of pretreatment (secondary facultative ponds). Important factors that affect primary facultative pond BOD and pathogen removal are areal loading, pond depth, detention time, temperature, solar radiation, quantity and quality of influent waste (Marais and Shaw, 1961).
There are an impressive number of design models available for the design of facultative ponds (Oswald and Gotaas, 1955; Marais and Shaw, 1961; Thirumurthi, 1969; Arceivala et al, 1970; McGarry and Pescod, 1970; Gloyna, 1971; Arthur, 1983; Mara et al, 1992b; and, Yanez, 1993). Some of the design formulae concentrate on principles involving fundamental interactions and relationships while others consider retention time and temperature. Generally, these design models tend to fall into one of three basic categories:

1. empirical model - based on permissible surface loading rate or using actual pond performance data.

2. kinetic model - assumed completely mixed reactor, with BOD following first-order removal kinetics.

3. dispersion model - based on hydraulic state of dispersion in pond being non-ideal.

Finney and Middlebrooks (1980) carried out an evaluation of empirical and kinetic design equations and found that predictions yielded by the design equations failed to agree with performance data. They concluded that consistent prediction of pond performance, without accurate projections of hydraulic residence time and consideration of climatic conditions, is impossible. A comprehensive review of the various design models was carried out by Ellis (1983).

The design model, based on permissible loading influenced by climate, generally recommended is that of Mara and Pearson (1987), which is given as:

\[
\lambda_s = 350(1.107 - 0.002T)^{1.25}
\]

where \( \lambda_s \) = surface BOD loading, kg/ha d

\( T \) = mean ambient air temperature, °C

It is expected since BOD removal in facultative ponds is the basis upon which they are designed that removal efficiencies reported in literature would be impressive. In general, this is so, however the removal ranges reported vary considerably. McGarry and Pescod (1970) collated and analysed data from primary facultative ponds operating under 143 different conditions and found that, for the most part, BOD removal is in the
range of 70-90%. Roesler and Preul (1970) reported between 60 and 90% BOD removal in facultative ponds in California. Bradley (1983) observed that a primary lagoon in Malaysia averaged 65% BOD removal. Facultative pond systems in Missouri and Kansas, USA were reported to have 60-92% BOD removal (Barerjl and Reuss 1987). Gray, (1989) reported that some facultative ponds achieve 90% BOD reduction. Removal of BOD in facultative ponds is dependent on the organic loading rate, temperature, and retention time.

Whereas anaerobic ponds digest biodegradable organic matter anaerobically, facultative ponds simultaneously operate anaerobically and aerobically but at different levels. There are 3 distinct layers within the 'ideal' facultative pond; upper aerobic, middle facultative, and bottom anaerobic. The oxygen found in the aerobic top layer is due to diffusion from the air and to photosynthetic algae which exists in a synergistic relation with aerobic bacteria. The photosynthetic action of the algal cells converts carbon dioxide into organic cell material with the liberation of oxygen:

\[ n\text{CO}_2 + n\text{H}_2\text{O} \rightarrow (\text{CH}_2\text{O})_n + \text{O}_2 \]  

Eq 2.3

The aerobic and facultative bacteria are able to metabolise the organic matter and reduce BOD by utilising dissolved oxygen and producing carbon dioxide, phosphates, and nitrates, which are in turn consumed by algal cells during their anabolic processes. At peak algal activity, the pH may rise as high as 10. This is as a result of algal activity removing carbon dioxide from solution quicker than it is being replaced by bacterial respiration. The resulting bicarbonate ion dissociation provides more carbon dioxide, and the alkaline hydroxyl ion which increases the pH value:

\[ \text{HCO}_3^- \rightarrow \text{CO}_2 + \text{OH}^- \]  

Eq 2.4

This essential relationship is illustrated by Fig 2.3:
As algal photosynthesis is related to the amount of solar radiation, the amount of oxygen produced will vary accordingly. Additionally, the concentration of oxygen and the pH value will change depending on the depth and the time of the day (Mara, 1976). During the non-daylight hours, no oxygen is produced. However, diffusion from atmospheric oxygen in a thin layer at the pond surface provides some oxygen for the continued respiration of algae and aerobic bacteria. In France, seasonal variations in the DO and pH were observed with the higher values being encountered in the warmer months (Troussellier et al, 1986).

Algae are inhibited by high hydrogen sulphide production derived from the sludge (Pearson et al, 1987c). It is believed that facultative ponds fail when there is excessive sulphide production associated with high organic loading (Silva, 1982). Due to the presence of algal and zooplankton biomass, this type of pond generally produces a final effluent that has a suspended solids concentration of >30 mg/l.

Sludge accumulation rates range from 10-90 mm per year according to Gray (1989). Other experiences with annual sludge accumulation rates in facultative ponds reported by Middlebrooks et al (1965) and Clare et al (1960) were 1.5-5.1 cm and 1.0-6.4 cm, respectively. Mara (1976) reported that facultative ponds require desludging after 10-15 years of use. The sludge at the bottom of facultative ponds is stabilised...
anaerobically. The soluble products of anaerobiosis in the sludge layer enter the pond liquid and are oxidised in the top layer (aerobic phase).

Table 2.3 illustrates the surface loading possibilities for various climates.

Table 2.3 Loading and design criteria for facultative ponds constructed in different climatic zones (Gloyna, 1971).

<table>
<thead>
<tr>
<th>Surface loading (BOD kg/ha day)</th>
<th>Population per ha</th>
<th>Retention time (days)</th>
<th>Environmental conditions</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt;10</td>
<td>&lt;200</td>
<td>&gt;200</td>
<td>Frigid zones, with seasonal ice cover, uniformly low water temperatures and variable cloud cover.</td>
</tr>
<tr>
<td>10-50</td>
<td>200-1000</td>
<td>200-100</td>
<td>Cold seasonal climate, with seasonal ice cover and temperate summer temperatures for short season.</td>
</tr>
<tr>
<td>50-150</td>
<td>1000-3000</td>
<td>100-33</td>
<td>Temperate to seem-tropical, occasional ice cover, no prolonged cloud cover.</td>
</tr>
<tr>
<td>150-350</td>
<td>3000-7000</td>
<td>33-17</td>
<td>Tropical, uniformly distributed sunshine and temperature, and no seasonal cloud cover.</td>
</tr>
</tbody>
</table>

It is clear from the Table 2.3 that facultative ponds are well suited to areas where there is sufficient solar radiation year round to enhance the self purification process as with warmer temperatures surface loading may be increased.

Although facultative ponds are designed primarily for BOD removal, pathogen removal as demonstrated by nematode reduction and faecal indicators die-off confirm that removal efficiencies are good. Mara and Silva (1986) carried out a study on experimental ponds in Brazil from which they concluded that effluents with \( \leq 1 \) egg/l can be produced in a pond system consisting of an anaerobic pond (1 day retention) followed by a facultative pond (5 days retention) and a maturation pond (5 days retention). They also reported 100% removal in a single facultative pond with 19 days retention time.

Many studies have reported excellent faecal coliform removals in facultative ponds. Moreno et al (1988) reported a 4 log removal in a pond in Spain. The mechanisms (include but are not limited to: temperature, starvation, solar radiation, predation, pH, algal derived antagonists, adsorption, sedimentation, and retention time), which are considered responsible for pathogen removal or reduction in facultative ponds have been researched and investigated in numerous studies since the early quarter of this century. As scientific research and debate continue to generate more data and more
complicated design equations, there remain many poorly explained phenomena occurring in the complex, dynamic pond systems. Although the laws of physics and chemistry govern behaviour in the world, modelling and accurately predicting pond behaviour appears to be elusive thus hindering rational design of ponds. This is due largely to poor understanding of hydraulic behaviour in ponds and hence their poor hydraulic design.

2.5.3 Maturation Ponds
Maturation ponds are characterised by having dissolved oxygen distributed throughout the pond liquor and are sometimes referred to as ‘aerobic ponds’. They are usually 1.0-1.5 m in depth with retention times varying from 3-15 days. Although biologically similar to facultative ponds, maturation ponds are distinguishable in that sludge accumulation, anaerobic digestion and sulphur transformation are minimal under normal operating conditions.

Maturation ponds are commonly used as a treatment method for improving or "polishing" the effluent from secondary biological processes. The improvement is normally in the reduction of pathogenic organisms demonstrated by faecal indicator bacteria, viruses, and protozoa. The use of these type of ponds is particularly important when considering the disposal of the final effluent. If agricultural reuse is to be considered, then maturation ponds are the most cost effective method by which to achieve the bacteriological effluent quality recommended by the Engleberg Report (IRCWD, 1985).

Toms et al (1975) in studies at the Rye Meads sewage treatment works in the UK reported that removal efficiency of BOD is low due to the initial low organic loadings on the maturation ponds. This is supported by Bradley (1983), who reported 39% removal, however removal as high as 92% was recorded by Cillie (1962) for maturation ponds treating humus tank effluent mixed with settled industrial wastes. It is practicable to assume that the organic loading to these maturation ponds was similar to that of facultative ponds in view of the addition of industrial wastes to the humus tank effluent. Roesler and Preul (1970) reported BOD removals of 68-96% in aerobic ponds in California.

The design of maturation ponds is based on faecal bacterial decay, not BOD removal as in anaerobic and facultative ponds. The first order rate constant or coefficient ($k_T$) for faecal coliform removal is recognised to be highly dependent on temperature. A wide range of values has been reported, however it is generally acceptable to calculate the $k_T$
for faecal coliform removal rate based on the equation proposed by Marais (1974) recognising that $k_T$ is highly temperature dependent he found that:

$$k_T = 2.6(1.19)^{T-20} \text{ Eq 2.5}$$

Where $T =$ mean ambient air temperature, °C

$k_T =$ first-order faecal coliform removal constant (d⁻¹)

Most pond designers use the lowest average air temperature of the year and apply the above equation. Upon obtaining a $k_T$ value, the retention time needed to achieve the desired level of faecal coliform removal is calculated based on the assumption of first order removal kinetics at temperature $T$°C, and if complete mixing is assumed, then:

$$N_e = \frac{N_i}{1 + k_T t} \text{ Eq 2.6}$$

Where

$N_e =$ number of faecal coliforms in effluent/100 ml

$N_i =$ number of faecal coliforms in influent/100 ml

$t =$ mean retention time in pond, days

The above equation is used for a single pond but as experience has shown multiple ponds in series are more efficient at faecal coliform removal thus Eq 2.6 becomes:

$$N_e = \frac{N_i}{\left[ (1 + k_{Ta} t_a)(1 + k_{Tf} t_f)(1 + k_{Tm} t_m)^n \right]} \text{ Eq 2.7}$$

Where

$N_e =$ number of faecal coliforms in final effluent/100 ml

$N_i =$ number of faecal coliforms in raw wastewater/100 ml

$k_T =$ first-order faecal coliform removal constant (d⁻¹)

$t =$ mean retention time in pond, days

$a =$ anaerobic pond

$f =$ facultative pond
Marais (1974) recommends a minimum acceptable retention time in maturation ponds of 3 days in order to minimise hydraulic short-circuiting.

Eq 2.5, although relatively simple, having two variables, has been used for many years to design maturation ponds. Notwithstanding it does not seem rational that the mechanisms controlling faecal coliform removal in pond environments are governed by air temperature alone. The equation shows that pond performance, as reflected by the $k_T$, is clearly dependent on climatic conditions. By taking into account the factors of organic loading and algal concentration, Polprasert et al (1983) attempted to re-define the design approach. The non-ideal dispersed flow equation of Wehner and Wilhelm (1956) has also been proposed for predicting bacterial removal instead of the first-order kinetics.

A number of factors have been identified or suggested as contributing to faecal bacterial reduction in maturation ponds. Some of the important factors are: high pH, high dissolved oxygen, germicidal action of sunlight, protozoal predation and nutrient starvation, and temperature (Kott, in press, Curtis, in press). Some studies (Caldwell, 1946) suggest that there is a close relation between pH and faecal coliform reduction with pH levels $\geq 10$ as being lethal. There have been some researchers (Moeller and Calkins, 1980) who are doubtful that pH is a significant factor in the survival of coliform bacteria in tertiary (maturation) lagoons because in their study, pH rarely exceeded 8.5 with excellent removals. Curtis (1990) investigated the photo-oxidation process occurring in waste stabilisation ponds and concluded that ‘light kills faecal coliforms in waste stabilisation ponds by an oxygen-mediated exogenous photosensitisation that interacts synergistically with elevated pH’. All of the mechanisms above are interlinked, therefore it is difficult to ‘partition’ the contribution of each removal mechanism in ponds.

Nevertheless most reports on faecal indicator bacteria in maturation ponds indicate excellent removals rates. Cillie (1962) reports 99.99% removal in maturation ponds treating humus tank effluent mixed with settled industrial wastes in 5:1 ratio. Mara et al (1992a) reports that in a ‘properly’ designed WSP system up to 5 logs or 99.999% faecal coliforms may be removed with time.
There is general agreement that sedimentation is the mechanism by which helminths are removed (Horan, 1990). Mara and Silva (1986) recommend to ensure adequate helminth removal, a minimum of two ponds with at least twenty days retention time be used. Ayres et al (1992) developed a design equation for helminth egg removal in ponds. Mara et al (1992a) recommends its use if the maturation ponds effluent is to be reused for irrigation.

Although enteric virus removal in maturation ponds is fairly well documented (Oragui et al, 1987; and Bausum et al, 1983) and favourably reported, none of the design methods reviewed attempted to include them. Horan (1990) reported that in his review of the limited data available pond systems with >30 days retention time should achieve at least a 4 log reduction in enteroviruses and a 3 log reduction in rotaviruses. From a virologic study conducted on a tertiary pond with 30 days retention in California by England et al (1967), only 1 log (91%) removal was reported. Rao et al (1981) reported that a full-scale system with 2.7-17.2 days retention, showed virus removal efficiencies ranging from 88-99% for the summer (30-35°C). They noted that a reduction in removal efficiency was experienced during the winter (20-26°C). Nupen (1970) reported 95% virus reduction in a series of 9 maturation ponds with 14 days retention time.

The mechanisms by which viruses are removed in waste stabilisation pond systems are not clear and the results vary widely due to methodological problems. Some studies point to good correlations between high removal and high pH (Funderburg et al, 1978; and Frederick and Lloyd, in press a) while others (Horan, 1990) point to adsorption to algae and other solids resulting in removal through subsequent sedimentation of the solids associated virus.

Nutrient removal has been excluded from this review as the main thrust is concerned with public health issues. However it is important to note that nutrients aid in the survival of pathogenic bacteria.

2.6 Effluent Disposal and Reuse
The effluent from ponds may be reused for various purposes due to the generally good quality in terms of BOD and pathogen. According to the World Bank (Newsweek, 1995) 80 countries representing 40% of the world's population are experiencing chronic water shortages. Consequently, the reuse possibilities of the effluent is an important consideration for choosing waste stabilisation pond treatment in many countries (Bartone and Arlosoroff, 1987).
In many arid and semi-arid countries, pond effluent reuse for agro-irrigation is an important part of water management. This is encouraged because effluent resulting from treatment in properly working ponds is generally of high quality - having very low pathogen content and BOD. In countries where the economy is dependent on tourism, sewage treatment in ponds is a practicable option because they are able to withstand the variation in loads between seasons and as a bonus suitable effluent may be reused for landscape or agro-irrigation.

The World Health Organisation (WHO, 1989) microbiological guidelines for the use of treated wastewater for irrigation are shown in Table 2.4:

Table 2.4 Microbiological quality guidelines for treated wastewater used for irrigation, (adapted from WHO, 1989).

<table>
<thead>
<tr>
<th>Reuse conditions</th>
<th>Exposed group</th>
<th>Intestinal nematodes$^a$</th>
<th>Faecal coliforms$^b$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unrestricted irrigation (crops likely to be eaten uncooked, sports fields, public parks)</td>
<td>Workers, consumers, public</td>
<td>≤ 1</td>
<td>≤ 1000$^c$</td>
</tr>
<tr>
<td>Restricted irrigation (cereal crops, industrial crops, fodder crops, pasture and trees$^d$)</td>
<td>Workers</td>
<td>≤ 1</td>
<td>No standard recommended</td>
</tr>
</tbody>
</table>

$^a$ Arithmetic mean no. of eggs per litre of *Ascaris lumbricoides*, *Trichuris trichiura* and the human hookworms.

$^b$ Geometric mean no. per 100 ml.

$^c$ A more stringent guideline (≤ 200 faecal coliforms cfu/100 ml) is appropriate for public lawns, such as hotel lawns, with which the public may come into direct contact.

$^d$ In the case of fruit trees, irrigation should cease two weeks before fruit is picked, and no fruit should be picked off the ground. Sprinkle irrigation should not be used.

These microbiological guidelines are based on in-depth review of epidemiological studies which showed that the excreted pathogens most prone to cause concerns in crop irrigation are the human intestinal nematodes and faecal bacteria (Shuval *et al*, 1986). These guidelines were revised from those recommended in 1973 (WHO, 1973) that were more stringent, faecal coliforms ≤100 cfu/100 ml).
It is interesting to note that during the same period that more stringent guidelines for irrigation were recommended by WHO, the following ‘less stringent’ guidelines were adopted in various countries:

1) Swimming (i.e. whole body immersion) was permitted in recreational waters containing up to 2000 faecal coliforms cfu/100 ml (Council of the European Communities, 1976).

2) River water containing up to 1000 faecal coliforms cfu/100 ml was allowed for irrigation in the United States, (US EPA, 1973).

3) Up to $10^5$ faecal coliforms cfu/100 g is allowed in food to be eaten raw, although the preferred is $<1000$ faecal coliforms cfu/100 g, (ICMSF, 1974).

The microbiological quality guidelines recommended are assumed to protect human health from risks associated with treated wastewater irrigation. WSPs are the wastewater treatment method most likely to achieve the faecal coliform guideline of 1000 cfu/100 ml (Bartone et al, 1985; and Mara et al, 1992b). However, the physico-chemical quality of the treated wastewater must also be considered as these affect the health and productivity of the plants. Generally the recommendations of the Food and Agricultural Organisation (FAO) are followed (Ayers and Westcot, 1985).

If WSPs are used to treat industrial wastewater then the presence of heavy metals and other toxic substances must be investigated. If the treatment system is receiving solely domestic wastewater then the main considerations are: electrical conductivity; sodium adsorption ratio; pH; nitrogen and boron (Mara et al, 1992a). Although the effluent BOD is not considered an important factor in assessing water quality for agricultural use, it will generally contain 60-90% algae (Cosser, 1982) which is a beneficial source of carbon, nitrogen and phosphorus to the soil (Bartone and Arlosoroff, 1987; and Mara et al, 1992b).

Other reuse possibilities for final pond effluent include; groundwater recharge, algae production, and fish and prawn production (Slack, 1974; Hepher et al, 1975; Goldman and Rhyther, 1976; and Reid, 1976). Generally, if agricultural reuse is not a consideration then the effluent may safely be discharged into receiving waters since the concentrations of physical, chemical and biological pollutants are normally reduced in pond treatment.
2.7 Summary
The following table (Table 2.5) compares conventional processes with waste stabilisation pond systems:

Table 2.5 Comparison of 2 conventional treatment processes and waste stabilisation ponds (adapted from Gray, 1989).

<table>
<thead>
<tr>
<th></th>
<th>Activated sludge*</th>
<th>Percolating filtration*</th>
<th>Waste stabilisation ponds**</th>
</tr>
</thead>
<tbody>
<tr>
<td>Capital cost</td>
<td>Medium</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td>Land area</td>
<td>Low - advantageous where land availability is restricted or expensive</td>
<td>Medium - requires 10 times the amount of land as activated sludge but less than WSPs</td>
<td>Large - requires many more times land area as activated sludge</td>
</tr>
<tr>
<td>Operating costs</td>
<td>High</td>
<td>Medium</td>
<td>Low</td>
</tr>
<tr>
<td>Influence of weather</td>
<td>Works well in wet weather, slightly affected by dry weather, less affected by low winter temperatures</td>
<td>Works well in summer but possible ponding in winter</td>
<td>Works well in tropical and temperate climates</td>
</tr>
<tr>
<td>Technical control</td>
<td>High: the microbial activity can be controlled; requires skilled and continuous operation</td>
<td>Little control possible except process modifications. Does not require continuous or skilled operation</td>
<td>Little control necessary. Does not require continuous or skilled operation</td>
</tr>
<tr>
<td>Nature of wastewater</td>
<td>Sensitive to toxic shocks, changes in loading, and industrial wastewaters; leads to bulking problems</td>
<td>Medium sensitivity to strong wastewaters, able to withstand changes in loading and toxic discharges</td>
<td>Able to withstand a wider range of wastewater strengths and shock loads than other conventional processes</td>
</tr>
<tr>
<td>Hydrostatic head</td>
<td>Small: low pumping requirement, suitable for site where available hydraulic head is limited</td>
<td>Large: site must provide natural hydraulic head otherwise pumping is required</td>
<td>Small: no pumping required unless effluent is recirculated</td>
</tr>
<tr>
<td>Nuisance</td>
<td>Moderate to low odour and fly problems. Noise may be a problem in both urban and rural areas</td>
<td>Moderate odour and severe fly problem probable. Quiet in operation</td>
<td>Low fly and odour problem if properly designed and SO$_2$ in wastewater &lt;300 mg/l</td>
</tr>
<tr>
<td>Final effluent quality</td>
<td>Poor nitrification but low in suspended solids except when bulking. High faecal coliform bacteria present</td>
<td>Highly nitrified, relatively high suspended solids. High faecal coliform bacteria present</td>
<td>Poor nitrification and may be high in suspended solids, Very low faecal coliform bacteria present</td>
</tr>
<tr>
<td>Secondary sludge</td>
<td>Large volume produced, high water content, difficult to dewater, less stabilised</td>
<td>Moderate volume produced, moderate water content, stabilised</td>
<td>Very small volume produced, lower water content, highly stabilised</td>
</tr>
<tr>
<td>Energy requirement</td>
<td>High: required for aeration, mixing and maintaining sludge floe in suspension and for recycling sludge</td>
<td>Low: natural ventilation, gravitational flow</td>
<td>Low: gravitational flow</td>
</tr>
<tr>
<td>Synthetic detergents</td>
<td>Possible foaming, especially with surfactants</td>
<td>Little foam</td>
<td>No foaming unless excessive surfactants</td>
</tr>
<tr>
<td>Robustness</td>
<td>Not very robust, high degree of maintenance on motors, not possible to operate without power supply</td>
<td>Sturdy, low degree of maintenance, possible to operate without power</td>
<td>Very robust, no power supply needed to operate</td>
</tr>
</tbody>
</table>

* data from Gray, 1989; ** author's addition, 1995

In Table 2.6, the characteristics of anaerobic, facultative and maturation ponds are compared:
Table 2.6 Comparison of anaerobic, facultative, and maturation pond characteristics (adapted from WHO, 1987b).

<table>
<thead>
<tr>
<th>MAIN CHARACTERISTICS</th>
<th>Anaerobic Pond</th>
<th>Facultative Pond</th>
<th>Maturation Pond</th>
</tr>
</thead>
<tbody>
<tr>
<td>Influent wastewater</td>
<td>domestic, some industrial</td>
<td>domestic, anaerobic pond effluent</td>
<td>effluent from 2° biological treatment (facultative ponds/conventional processes)</td>
</tr>
<tr>
<td>Effluent</td>
<td>grey, few settleable solids, 50-70% BOD reduction, septic odour</td>
<td>green, few settleable solids, 60-90% BOD reduction, odourless</td>
<td>pale green, very little settleable solids, odourless</td>
</tr>
<tr>
<td>Dissolved oxygen</td>
<td>absent</td>
<td>present</td>
<td>present</td>
</tr>
<tr>
<td>Retention time</td>
<td>1-5 days</td>
<td>7-50 days</td>
<td>3-15 days</td>
</tr>
<tr>
<td>Water depth</td>
<td>2.5-5.0 m</td>
<td>1.5-2.0 m</td>
<td>1.0-1.5 m</td>
</tr>
<tr>
<td>Effluent pH</td>
<td>6.5-7.5</td>
<td>7.0-10.0</td>
<td>7.8-10.5</td>
</tr>
<tr>
<td>Algae in effluent</td>
<td>absent</td>
<td>present</td>
<td>few present</td>
</tr>
<tr>
<td>Coliforms in effluent</td>
<td>many (10^6)</td>
<td>considerable (10^6)</td>
<td>some (10^2-10^3)</td>
</tr>
</tbody>
</table>

Although a pond may be classified at the design stage, it may operate in other modes depending on operational parameters. For example, maturation ponds can become facultative or anaerobic depending on load factors and other biochemical processes (Parker, 1979).

The various reported removal efficiencies for BOD reduction, pathogen indicator removal and sludge accumulation in waste stabilisation ponds are summarised in Table 2.7.
Table 2.7 A summary of removal efficiency for BOD reduction, pathogen indicator removal and for sludge accumulation in waste stabilisation ponds.

<table>
<thead>
<tr>
<th>Performance Parameter</th>
<th>Anaerobic Pond</th>
<th>Facultative Pond</th>
<th>Maturation Pond</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Faecal coliform % removal</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>99.97, 1 d ret., Brazil (Oragui et al, in press)</td>
<td>99.6, 61 d ret., Brazil (Ceballos et al, in press)</td>
<td>99.999, South Africa (Cillie, 1962)</td>
<td></td>
</tr>
<tr>
<td>75, 0.6 d ret., Kenya (Alabaster et al, 1991)</td>
<td>94.9, 4-5 d ret., Spain (Soler et al, in press)</td>
<td>99.7, 7 d ret., South Africa (Gaillard &amp; Crawford, 1964)</td>
<td></td>
</tr>
<tr>
<td>89.5, 4.6 d ret., Kenya (Grimason et al, in press)</td>
<td>99.7, 22.42 d ret., Kenya (Alabaster et al, 1991)</td>
<td>89.5, 3 d ret., Kenya (Grimason et al, in press)</td>
<td></td>
</tr>
<tr>
<td>87-100, 8 d ret., 2 ponds, Jordan (Pescod, in press)</td>
<td>84.1, 14 d ret., Kenya (Alabaster et al, 1991)</td>
<td>66-79, 3 d ret., Cayman Islands (Frederick, in this study)</td>
<td></td>
</tr>
<tr>
<td><strong>Helminth, % removal</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>26.6, 10hr ret., Kenya (Ayers et al, 1993)</td>
<td>100, 61 d ret., Brazil (Ceballos et al, in press)</td>
<td>100, 20 d ret., Brazil (Dixo et al, in press)</td>
<td></td>
</tr>
<tr>
<td>87-100, 8 d ret., 2 ponds, Jordan (Pescod, in press)</td>
<td>99.999, 3 d ret., South Africa (Jagals &amp; Lues, in press)</td>
<td>100, 5.5 d ret., Peru (Yanez et al, 1980)</td>
<td></td>
</tr>
<tr>
<td>99.999, 3 d ret., South Africa (Jagals &amp; Lues, in press)</td>
<td>98, 8.5 d ret., Kenya (Grimason et al, in press)</td>
<td>99.9999, 3 d ret., South Africa (Jagals &amp; Lues, in press)</td>
<td></td>
</tr>
<tr>
<td>33, 0.8 d ret., Kenya (Grimason et al, in press)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>BOD reduction %</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>80, 2-3 d ret., Yemen (Veenstra et al, in press)</td>
<td>95, 61 d ret., Brazil (Ceballos et al, in press)</td>
<td>92, South Africa (Cillie, 1962)</td>
<td></td>
</tr>
<tr>
<td>53, 5 d ret., Jordan (Saqqar &amp; Pescod, in press)</td>
<td>45, with 15 d ret., Yemen (Veenstra et al, in press)</td>
<td>39, 4.1 d ret., Malaysia, (Bradley, 1983)</td>
<td></td>
</tr>
<tr>
<td>83, 1 d ret., Brazil (Oragui et al, in press)</td>
<td>90, 135 d ret., (3 ponds) Surinam (Ramkisar, 1986)</td>
<td>67, 7 d ret., South Africa (Gaillard &amp; Crawford, 1964)</td>
<td></td>
</tr>
<tr>
<td>47.1-82.2, 2.5 d ret., Australia (Parker et al, 1950)</td>
<td>57-60, 8 d ret., Cayman Islands (Frederick, in this study)</td>
<td>77.1, 17.5 d ret., Australia (Mitchell &amp; Williams, 1982)</td>
<td></td>
</tr>
<tr>
<td>46, 8 d ret., Spain (Soler et al, in press)</td>
<td>47, 1, 26 d ret., Kenya (Grimason et al, in press)</td>
<td>11-29, 3 d ret., Cayman Islands (Frederick, in this study)</td>
<td></td>
</tr>
<tr>
<td>59, 5.3 d ret., Jordan (Al-Salem &amp; Lumbers, 1987)</td>
<td></td>
<td>82.6, 3 d ret., Kenya (Grimason et al, in press)</td>
<td></td>
</tr>
<tr>
<td>50, 0.6 d ret., Kenya (Alabaster et al, 1991)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>67, 6.6 d ret., Kenya (Alabaster et al, 1991)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>34.6, 4.6 d ret., Kenya (Grimason et al, in press)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Sludge accumulation</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>40 cm in 2 years, 2.5 d ret., Australia (Parker et al, 1950)</td>
<td>0.585 cm/yr, France, (Schertrie &amp; Racault, in press)</td>
<td>12 cm in 7 years, Cayman Islands (Frederick, in this study)</td>
<td></td>
</tr>
<tr>
<td>2-4 cm/yr, India, (Dave &amp; Jain, 1967)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1.7 m in 44 months, Jordan (Pescod, in press)</td>
<td>14.5 cm in 5 years, 43 d ret., France, (Racault, 1993)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>34 cm in 7 years, Cayman Islands (Frederick, in this study)</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
It is clear from the preceding review that WSPs offer considerable advantages over conventional treatment and it was with these advantages in mind that the WSP system in the Cayman Islands was designed. The system commissioned in February 1988 is described in the following chapter.
CHAPTER 3

3.0 DESIGN AND OPERATION OF CAYMAN WASTE STABILISATION POND SYSTEM

3.1 Physical Description of Treatment System
In a "conventional" sewage treatment lagoon system, an anaerobic pond is generally chosen as the primary stage of treatment. However, the Water Authority-Cayman's decision to omit the anaerobic treatment stage was based on several considerations.

The first, and major concern was the possibility of odour emission due to incomplete anaerobiosis occurring in anaerobic ponds (Mara, 1976). As the treatment system is located less than 0.8 km from the West Bay Beach tourist area downwind, the potential risk of odour was a serious concern. Some authors recommend that anaerobic ponds should be located at least 1 km away from the nearest residence (Middlebrooks et al, 1982 and WHO, 1987b).

Secondly, some authors suggest that anaerobic ponds are unnecessary if the sewage to be treated is weak (Gray, 1989). Anaerobic ponds are best suited where there are discharges from industries into the system as these ponds are able to partially stabilise strong industrial wastewater (Ellis, 1983). The influent of the WAC sewerage system was expected to be relatively weak as it served a tourist area and as pointed out in Chapter 1, tourists use almost twice the amount of water as a person in a typical Caymanian household. The strength of septage was unpredictable, however, its representation in the influent was expected to be <5% of the total flow to the works. Additionally, it was to be mixed with a portion of the final effluent from the last maturation pond recirculating back into the works thereby reducing the shock that it may have on the ponds. The Cayman Islands does not have an industrial economic base, thus the possibility of toxic chemicals entering the sewerage system was low.

Another disadvantage of using anaerobic ponds is that they require desludging earlier (3-5 yrs) than facultative ponds. One implication of desludging is that it is necessary to take the pond out of service, therefore unless 2 anaerobic ponds are available, the next facultative pond in the series would be required to receive the influent during the period of desludging.
In view of the above considerations, the decision was made to design a sewage treatment pond system without anaerobic lagoons. The treatment system consists of 2 facultative and 2 maturation ponds. The ponds are situated on reclaimed mangrove swampland (Fig 3.1) located southeast of the West Bay Beach area (Fig 1.4, pg 11).

The bottom of the ponds are 3 m above the ground level and are lined with non-woven geotextile (100 mil) which is then covered with an impermeable high density polyethylene (HDPE) plastic, 60 mil thick (Fig 3.2). Lining and raising above the ground level were necessary because of the low elevation above mean sea-level of the mangrove area.

The treatment site is enclosed by a 2.4 m high chain-link fence with warning signs posted to discourage public trespassing. Fig 3.3 is a view of the system from the air showing aerators in operation.
Fig 3.2 Facultative pond 1.1 during installation of lining (geotextile underlay shown in left foreground).

Fig 3.3 Aerial view of sewage treatment ponds in Grand Cayman, Cayman Islands, taken in 1990. (Aerators in operation in both facultative ponds and baffles' effects seen in both maturation ponds).
3.2 Design Assumptions

In order to arrive at design figures for the treatment lagoons, assumptions concerning sewage strength, population growth, and commercial development were made by the design engineers. Since this project was the first on the island to provide a public wastewater collection system, there were limited data available on local sewage strength. However, as the hotels were using package sewage treatment plants, samples were taken of the influents and effluents at selected plants and analysed by the WAC for BOD, suspended solids (SS) and electrical conductivity (EC). Subsequently, design figures were assumed from the results obtained in addition to consideration of typical United States statistics for water consumption by tourist and residential development.

The flow from the area to be sewered was estimated at 1200 m$^3$/day based on hotels' occupancy rates and water consumption in 1985. The system was designed for a maximum flow of 2813 m$^3$/day in 1996 after which the treatment plant would be expanded further with 2 additional maturation ponds. At the projected growth rates and with the 2 extra ponds, it was expected that the system would be able to adequately treat up to 4543 m$^3$/day sewage until the year 2006.

3.2.1 Design of the Facultative Ponds

It was the intention of the WAC to sell the final effluent for reuse as irrigation water to a nearby golf club. The opportunity for revenue (US$12,000.00 per month) to be earned from the effluent of the treatment process was an integral part of the design. The effluent standards in the Water Authority Regulations (WA Regulations, 1985) are similar to the recommendations of the 8th report of the Royal Commission on Sewage Disposal which dates back to 1912. The Authority's regulations stipulate that all domestic wastewater effluents must meet 30/30 standards before discharge. The 30 mg/l BOD and 30 mg/l SS, do not give any estimation of health risk if human contact is made with sewage effluents. These parameters were chosen as maximum levels of organic matter that could be discharged into rivers without fish-kills and other signs of severe organic pollution. These standards were adopted even though, the WAC allows effluent discharge only to deepwells, not to marine outfalls (Chapter 1).

Because reuse was proposed and the discharge standard in the WA law does not address human or environmental health risk, it was decided to adopt the California State microbiological standards (faecal coliforms: 100 cfu/100 ml, Anon., 1978) for recreational reuse in addition to the Royal Commission standards. Therefore, the sewage treatment system was designed to achieve these levels in the final effluent.
As described in Chapter 2, the design of facultative ponds may be based on several approaches. In the Cayman Islands, the design model used for calculation of BOD surface loading of the facultative waste stabilisation ponds was mainly based on the empirical method of Arthur (1983). Effluent bacterial strength was calculated based on first order kinetic rates assuming complete mixing.

The resulting assumptions for the facultative ponds design are presented in Table 3.1:

Table 3.1  Design assumptions for the design of facultative waste stabilisation ponds in Grand Cayman (van Zanten, 1987).

<table>
<thead>
<tr>
<th>DESIGN PARAMETER</th>
<th>ASSUMPTION</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature (lowest monthly average)</td>
<td>24°C (ambient air)</td>
</tr>
<tr>
<td>Maximun BOD loading rate</td>
<td>420 kg/ha d</td>
</tr>
<tr>
<td>Influent volume</td>
<td>2813 m³/d (1996)</td>
</tr>
<tr>
<td>BOD strength of influent</td>
<td>205 mg/l</td>
</tr>
<tr>
<td>Faecal bacterial concentration of influent</td>
<td>9.28 x 10⁶ cfu/100 ml</td>
</tr>
</tbody>
</table>

The design flows for 1987 were estimated to be 1800 m³/day for a population of 4460. There is provision for 2 additional ponds to be built in 1996 to provide treatment of 4543 m³/day until the year 2006. Included in the design flow is allowance for ≤5% septage disposal, 5% groundwater infiltration into sewers, and 20% recirculation of the final effluent from maturation pond 2.2.

As a result of the assumptions in Table 3.1, the following physical parameters (Table 3.2) were determined for the facultative lagoons based on the use of empirical design equations in order to achieve the effluent quality required:
Table 3.2 Physical properties of the Cayman Islands facultative waste stabilisation ponds used at the design stage (van Zanten, 1987).

<table>
<thead>
<tr>
<th>PARAMETER</th>
<th>PHYSICAL DESCRIPTION</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number of facultative ponds</td>
<td>2 (in parallel)</td>
</tr>
<tr>
<td>Width of facultative ponds</td>
<td>61 m (@ TWL)</td>
</tr>
<tr>
<td>Length of facultative pond</td>
<td>167 m (@ TWL)</td>
</tr>
<tr>
<td>Nominal area of the 2 ponds</td>
<td>20374 m² (@TWL)</td>
</tr>
<tr>
<td>Depth of each pond</td>
<td>2.0 m</td>
</tr>
<tr>
<td>Embankment slope</td>
<td>1 in 2</td>
</tr>
<tr>
<td>Nominal volume of the 2 ponds</td>
<td>37164 m³</td>
</tr>
<tr>
<td>Average retention (1996)</td>
<td>13.21 days</td>
</tr>
</tbody>
</table>

*note: TWL = top water level*

Using the physical design properties proposed in Table 3.2, the facultative lagoons were expected to reduce unfiltered influent BOD by 80% (to 35 mg/l) and influent faecal coliform by >98% (to $1.4 \times 10^5$ cfu/100 ml) by the time the flow reached that used in the design (van Zanten, 1987).

3.2.2 Design of Maturation Ponds

The design of the maturation ponds was based on bacterial decay as their major function is the reduction of pathogens. The 2 most important factors in bacterial reduction models are retention time and the die-off constant or coefficient ($k_T$) which is temperature dependent and was described in the previous chapter. The $k_T$ value applied in the design of the maturation ponds was $5.21 \, \text{d}^{-1}$ obtained from Eq 2.5 (pg 33) using the minimum ambient air temperature of 24°C.

It was determined that the maturation ponds should be built according to the specifications in Table 3.3 in order to continue to meet the desired microbiological effluent standards until the year 1996.
Table 3.3 Physical parameters of the Cayman Islands maturation waste stabilisation ponds at the design stage (van Zanten, 1987).

<table>
<thead>
<tr>
<th>PARAMETER</th>
<th>PHYSICAL DESCRIPTION</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number of maturation ponds</td>
<td>2 (in series, with baffles)</td>
</tr>
<tr>
<td>Width of maturation ponds</td>
<td>60 m (@ TWL)</td>
</tr>
<tr>
<td>Length of maturation pond</td>
<td>109 m (@ TWL)</td>
</tr>
<tr>
<td>Area of each pond</td>
<td>6540 m² (@TWL)</td>
</tr>
<tr>
<td>Depth of each pond</td>
<td>1.75 m</td>
</tr>
<tr>
<td>Nominal volume of each pond</td>
<td>11445 m³</td>
</tr>
<tr>
<td>Nominal retention time of each pond (1996)</td>
<td>4.03 days</td>
</tr>
<tr>
<td>Cumulative nominal retention time of WSP system (1996)</td>
<td>21.28 days</td>
</tr>
</tbody>
</table>

*note: TWL = top water level*

The resulting physical design parameters (Table 3.3) for the maturation ponds of the WAC’s treatment system were designed to provide a final effluent BOD of 12 mg/l and faecal coliform population of $1 \times 10^2$ cfu/100 ml which would be suitable for irrigation on a recreational area (van Zanten, 1987).

3.3 Routine Operation of the WSP Treatment System

The configuration of the treatment system designed was 2 facultative ponds operating in parallel followed by 2 maturation ponds (baffled) in series. Presently, and since commissioning in 1988, the facultative ponds 1.1 and 1.2, receive equal volumes of incoming raw sewage.

From these 2 facultative ponds, the flow enters the primary maturation pond 2.1 and then flows on to the final maturation pond 2.2. The direction of flow throughout the system is shown in Fig 3.4.
3.3.2 Flow through the WSP System

The incoming flow is pumped up to the inlet works channel (Fig 3.5) where it passes through a bar screen for removal of gross solids. From there it passes on to a grit...
chamber. The grit chamber, utilising a Mark 11 Pista Grit Trap serves to remove solid particles such as pebbles, corn and beans. Problems were experienced with the grit trap and it was out of order for a period of 12 months between 1992 and 1993.

Fig 3.5 The inlet works at the WAC waste stabilisation ponds treatment system.
The incoming flow to the system was originally measured by a Steven's Total Flow meter gauge located in the standing wave channel with logging onto a chart recorder. A reliable flow measuring device is necessary in order to determine organic loadings and evaporation losses (Gloyna, 1971) as well as the data necessary to determine when capacity of the system must be increased. It follows that appreciable errors in the interpretation of monitoring data could result if the flow measurement device fails to operate reliably.

Numerous problems were experienced with the Steven's flow meter. Disintegration of the delicate mechanisms due to the corrosiveness of the incoming sewage resulted in extended periods of no measurements. Additionally, due to hydraulic jumps, the accuracy of the recorded flow was questionable. In February, 1994 a Doppler flow meter was installed to measure the inflow to the works (including recirculation). The WAC New Works engineer, by comparing the Steven's Gauge flow meter measurements simultaneously with the Doppler meter for approximately 330 days and by using the hours the pumps were running at the final pumping station (PS#1) to the inlet works, and knowing their specifications, was able to apply a correction factor to all flow measurements taken prior to the installation of the new meter. The good correlation of $R^2 = 0.77$ between the corrected Steven’s gauge flows and those of the Doppler meter are illustrated in Fig 3.6. The data plotted represents 328 data pairs.

Fig 3.6 Correlation between Doppler flow meter measurements and Steven's gauge flow data corrected using the number of the hours pumps were run (van Zanten, 1995).
Measurements of final effluent flow from the initially installed Kent water meter were not accurate as it is unable to operate normally when fluids passing through have suspended material (McTaggart, 1990). Suspended material interferes with the turning mechanism in the meter, causing turbulence which makes it revolve faster, resulting in falsely elevated measurements. As this was unacceptable, the WAC discontinued the use of the Kent meter and will be obtaining another Doppler flow meter for measurement of the final effluent in the near future.

3.3.3 Inlet and Outlet Configurations
The inlets to the two facultative ponds are submerged and each inlet consists of two 0.30 m diameter pipes located 0.80 m (invert level) from the bottom of the ponds set at 90° from each other. These pipes extend 1.0 m into the northern end of both ponds. Fig 3.7 shows facultative pond 1.1 prior to it being commissioned in February 1988. The outlet arrangement of the facultative ponds during construction is shown in Fig 3.8.

The outlet configurations of the 2 facultative ponds each consists of 2 submerged 0.30 m diameter pipes located 0.35 m (invert level) from the bottom of the ponds at 60° from each other (Fig 3.9). These pipes extend <1.0 m into the southern end of ponds. Effluent flows into and over a weir chamber and from there into maturation pond 2.1.

The inlet configuration to the maturation ponds are similar to that of the facultative ponds except for the distance of the pipes from the pond bottom. In maturation pond 2.1, the distance of the inlet pipe from the bottom is 0.70 m (invert level), and in maturation pond 2.2, the distance is 0.50 m (invert level).

The outlets of the maturation ponds are also similar to that of the facultative ponds except that the pipes are located 0.50 m (invert level) from the bottom of maturation pond 2.1, and 0.45 m (invert level) from the bottom in maturation pond 2.2.
Fig 3.7 Facultative pond 1.1 prior to commissioning in February, 1988, showing inlet pipes from the inlet works.

Fig 3.8 Outlet configuration from facultative ponds shown during construction.
Fig 3.9 Schematic of the inlet outlet arrangement connecting facultative pond 1.1 and maturation ponds 2.1. (Pipe shown on the top of weir chamber receives the inflow of effluent from facultative pond 1.2)

3.3.4 Baffles in Maturation Ponds
Baffles were installed in the maturation ponds in order to reduce the possibility of short-circuiting and ensure increase pathogen removal efficiency. The baffle consists of a HDPE floating curtain stretched across the width of the maturation ponds and held in place by lead weights on the bottom and steel wires at the top. There are 4 windows (0.40 m x 0.90 m) through which the pond water flows. These are located as shown in schematic drawing Fig 3.10.

Fig 3.10 Schematic cross section of baffle installed across width of maturation ponds (not to scale).
A view of maturation pond 2.1 with baffles in place is shown in Fig 3.11.

3.3.5 Septage Treatment, Effluent Disposal and Recirculation
Septage from septic tank emptiers is mixed in the wet well with recirculated final effluent before entering the facultative ponds. This serves to dilute the strong septic mixture, thereby reducing its potential shock to the treatment system. Presently septage flows represent <1% of the total flow to the WSP system. A typical tanker is shown emptying its load in Fig 3.12.
Fig 3.12 Septage being received at the WSP system in Grand Cayman.

It is estimated that 20% of the final effluent is recirculated back to the facultative ponds. As at present the volume of effluent recirculated is not measured with a flow meter, this assumption is made based on the hours the pumps run each day. The remaining final effluent is disposed of into 2 deepwells (Fig 3.13) with depths of 50 m.
3.3.6 Aerators in Facultative Ponds
In late 1989 eight electric motor driven propellor aspirator aerators were installed in each facultative pond (see Fig 3.14). These aerators work on the principle that the propellor spins quickly, creating a vacuum that sucks atmospheric air below the surface of the water. The flow pattern created by the propellor is specified by the manufacturer to be horizontal rather than vertical. It was necessary to install the aerators due to high
levels of hydrogen sulphide production caused by high sulphate levels in the incoming sewage. The cause and effects of the sulphur transformations occurring in the WSP system are further discussed in Chapter 8.

The aerators were operated when odour production was offensive to neighbours downwind of the ponds. This was an unsatisfactory situation because operating the aerators continuously increased energy costs immensely (Fig 3.15). Four of the aerators in each pond were used continuously for 3 months for 24 hours per day (Tables 3.4 and 3.5).

Table 3.4 History of aerators' use in facultative pond 1.1.

<table>
<thead>
<tr>
<th>DATE</th>
<th>AVERAGE hours/day</th>
<th>NUMBER OF AERATORS ON</th>
</tr>
</thead>
<tbody>
<tr>
<td>13 Oct89-13 Jan90</td>
<td>24 hrs</td>
<td>4</td>
</tr>
<tr>
<td>14 Jan90-9 May90</td>
<td>16 hrs @ night</td>
<td>4</td>
</tr>
<tr>
<td>10 May90-27 Nov91</td>
<td>none</td>
<td>off</td>
</tr>
<tr>
<td>28 Nov91-15 Mar92</td>
<td>15 hrs @ night</td>
<td>4</td>
</tr>
<tr>
<td>16 Mar92-21 Jan93</td>
<td>none</td>
<td>off</td>
</tr>
<tr>
<td>22 Jan93-26 Mar93</td>
<td>17 hrs @ night</td>
<td>8</td>
</tr>
</tbody>
</table>

Table 3.5 History of aerators' use in facultative pond 1.2

<table>
<thead>
<tr>
<th>DATE</th>
<th>AVERAGE hours/day</th>
<th>NUMBER OF AERATORS ON</th>
</tr>
</thead>
<tbody>
<tr>
<td>13 Oct89-13 Jan90</td>
<td>24 hrs</td>
<td>4</td>
</tr>
<tr>
<td>14 Jan90-1 Apr90</td>
<td>16 hrs @ night</td>
<td>4</td>
</tr>
<tr>
<td>2 Apr90-27 Nov91</td>
<td>none</td>
<td>off</td>
</tr>
<tr>
<td>28 Nov91-15 Mar92</td>
<td>15 hrs @ night</td>
<td>4</td>
</tr>
<tr>
<td>16 Mar92-21 Jan93</td>
<td>none</td>
<td>off</td>
</tr>
<tr>
<td>22 Jan93-1 Mar93</td>
<td>24 hrs</td>
<td>6</td>
</tr>
</tbody>
</table>

As the increased costs were undesirable, a trial period during which the aerators were turned off during the day and on at night began in January 1990. In April 1990, the aerators were turned off completely until November 1991, and were then used until
March 1992. After approximately 10 months of being turned off, the aerators were used for 2 months starting in January 1993 operating at 24 hours per day. At the end of March 1993, all of the aerators were disassembled and removed from the ponds.

Fig 3.14 Electric motor driven propellor aspirator aerator prior to installation into facultative pond.
3.4 Evaluation of WSP System's Operational Performance

Monitoring of a WSP system may be carried out at one of 3 levels depending on the local situation (Mara and Pearson, 1987; and Pearson et al, 1987a). The first level is the monthly analysis of the final effluent to ensure that effluent discharge or reuse standards are being met. The second level is required in order to assess the entire pond system and involves routine samples from each treatment pond and the incoming wastewater when failures in final effluent are detected in level 1. The third level of assessment is an intensive, time consuming and expensive process and involves in-pond sampling of a variety of supplementary parameters. However it is only through this type of research that the pond design can be adapted to local conditions thus it is a very beneficial and productive activity. The WSP evaluation system undertaken through this study contained the elements of level 2 and level 3 that were considered to be appropriate for the Cayman Islands situation. The routine monitoring methods are described in the next chapter.
CHAPTER 4

4.0 ROUTINE MONITORING METHODS

4.1 Justification for Establishing WSP Monitoring Programme in the WAC

As the final effluent was to be sold, a basic monitoring programme was developed and implemented by the author, on behalf of the WAC, in March, 1988. This programme was aimed at determining and assessing the general performance of the system and whether the final effluent met the standards required for reuse. A very important parameter of this monitoring programme was the measurement of salinity, important because the effluent would be used to irrigate golf-course grass which could not survive >3500 µS/cm salinity (measured as electrical conductivity). Additionally, 35% of the sewers and all of the interceptors are situated below the groundwater table which in this area of the island is very saline (40000 µS/cm), approaching the salinity of seawater (54000 µS/cm). Therefore, in order to ascertain the integrity of the sewers with regard to groundwater infiltration regular scrutiny of the salinity of the influent was necessary.

4.1.2 Rationale for Selection of Analytical Methods

Analysis of sewage has traditionally been confined to a limited number of parameters. The most important of these are biochemical oxygen demand (BOD), chemical oxygen demand (COD), suspended solids (SS), ammonia (NH₃), nitrates (NO₃), phosphates (PO₄); because they profoundly affect the water quality and biota of receiving water bodies. The growing awareness of health risks associated with pathogens and parasites in sewage during the last 20 years has stimulated the development and application of microbiological and biological monitoring methods. This is particularly relevant to lagoon treatment where it has become clear that the removal efficiency for the aetiological agents of disease are in general far higher than that found in the so-called 'conventional' sewage treatment processes.

The aim in the collection and analysis of information on the physico-chemical and microbiological behaviour of ponds is to assist in understanding the complex interactions which govern the treatment process and help correct malfunctioning systems. Additionally it is necessary to monitor performance and assess whether discharge standards are being met.
In a research study of the type described here it was clear from the outset that considerable care had to be exercised in the selection and frequency of parameters for study. Advice, in this respect, was sought from Pescod (1989), Mara (1990) and in Pearson et al (1987a). These experts suggested augmenting the monitoring programme by the addition of ova/parasite surveys, chlorophyll $a$, retention time studies, and sludge depth measurements.

The WAC laboratory serves as a quality control centre for the operations of the Authority. It was staffed prior to July 1994, by the author and a laboratory technician. It is presently staffed by a senior laboratory technologist and a laboratory technician. Besides research work on the waste stabilisation ponds, routine analyses for intensive groundwater monitoring and quality control of piped water supplies are carried out.

Budgetary constraints prohibited the purchase of elaborate, sophisticated equipment, resulting in several types of analyses being carried out with test-kits utilising procedures based on the Standard Methods (APHA, 1985). Where applicable, control standards are used routinely during sample analyses. A monitoring programme was established in March, 1988, by the author at the time the ponds were being filled. This programme contained the basic parameters for monitoring of sewage lagoons. As it became clear that different types of analyses were necessary to improve the understanding of the treatment process, new and varied determinands were added. The programme was adjusted as shown in Table 4.1.

Grab samples of the ponds' effluents were collected at the overflow chamber as indicated in Fig 4.1 and Fig 4.3. This was considered a representative sample of the pond effluents as it is from the outlet of the pond. Influent grab and 24 hour composite samples (with refrigeration) were collected as indicated in Figs 4.2 and 4.3. Septage samples are collected and composited for the 8 hour day (there are no night discharges) by the sewage plant operator as tankers are discharging into the wet-well.

All samples were collected and analysed weekly up until July 1992 after which samples were collected every fortnight. All samples for the routine monitoring programme are collected from the sewage treatment works between 0900 hours and 1000 hours in the morning.
Table 4.1 Water Authority-Cayman Laboratory, Sewage Treatment Works Monitoring Programme, 1991.

<table>
<thead>
<tr>
<th>Determinand</th>
<th>Method</th>
<th>Frequency</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Microbiological Analyses:</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>faecal coliform, cfu/100 ml</td>
<td>membrane filtration</td>
<td>weekly</td>
<td>influent, septage</td>
</tr>
<tr>
<td>ova/parasite, no./l</td>
<td>membrane filtration</td>
<td>monthly for 4 months (discussed in Chap. 9)</td>
<td>influent &amp; pumping station 1</td>
</tr>
<tr>
<td>phyto/zooplankton, no./l</td>
<td>Neubauer cell</td>
<td>monthly for 4 months (discussed in Chap. 9)</td>
<td>pond effluents</td>
</tr>
<tr>
<td><strong>Chemical Analyses:</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(mg/l, unless otherwise indicated)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ammonia</td>
<td>Nesslerisation</td>
<td>weekly</td>
<td>influent, septage, ponds</td>
</tr>
<tr>
<td>BOD₅ unfiltered/filtered &amp; DO</td>
<td>Winkler, electrode</td>
<td>weekly</td>
<td>influent, septage, ponds</td>
</tr>
<tr>
<td>chloride</td>
<td>argentometric</td>
<td>monthly</td>
<td>influent, septage, ponds</td>
</tr>
<tr>
<td>COD unfiltered/filtered</td>
<td>dichromate reflux</td>
<td>weekly</td>
<td>influent, septage, ponds</td>
</tr>
<tr>
<td>chlorophyll a µg/l</td>
<td>spectrophotometric</td>
<td>monthly for 4 months (discussed in Chap. 9)</td>
<td>all ponds</td>
</tr>
<tr>
<td>hydrogen sulphide</td>
<td>methylene blue</td>
<td>weekly</td>
<td>influent, all ponds</td>
</tr>
<tr>
<td>sulphate</td>
<td>turbidimetric (barium chloride)</td>
<td>weekly</td>
<td>influent, septage, ponds</td>
</tr>
<tr>
<td><strong>Physico-Chemical:</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>colour</td>
<td>visually observed</td>
<td>daily</td>
<td>influent, all ponds</td>
</tr>
<tr>
<td>electrical conductivity µS/cm</td>
<td>electrometric</td>
<td>weekly</td>
<td>influent, septage, ponds</td>
</tr>
<tr>
<td>odour</td>
<td>nasal (subjective)</td>
<td>daily</td>
<td>influent, all ponds</td>
</tr>
<tr>
<td>pH units</td>
<td>potentiometric</td>
<td>weekly</td>
<td>influent, all ponds</td>
</tr>
<tr>
<td>sludge depth in cm</td>
<td>white towel</td>
<td>bi-annually</td>
<td>all ponds</td>
</tr>
<tr>
<td>temperature in °C</td>
<td>Hg in glass</td>
<td>weekly</td>
<td>influent, all ponds</td>
</tr>
<tr>
<td>total dissolved solids</td>
<td>electrometric</td>
<td>weekly</td>
<td>influent, all ponds</td>
</tr>
<tr>
<td>total fixed solids</td>
<td>gravimetric</td>
<td>weekly</td>
<td>influent, all ponds</td>
</tr>
<tr>
<td>total suspended solids</td>
<td>gravimetric</td>
<td>weekly</td>
<td>influent, all ponds</td>
</tr>
<tr>
<td>total volatile solids</td>
<td>gravimetric</td>
<td>weekly</td>
<td>influent, all ponds</td>
</tr>
</tbody>
</table>
Fig 4.1 Collection of pond effluent samples at weir chamber.

Fig 4.2 24-hour composite sample collection apparatus.
4.2 Microbiological Analysis

4.2.1 Faecal Coliform Analysis

The indicator bacteria used in this study were faecal coliforms (see Chapter 9). The faecal coliform group are Gram-negative, thermotolerant, non-sporing rods that are capable of aerobic and facultatively anaerobic growth in the presence of bile salts, are oxidase-negative, and are able to ferment lactose with the production of acid and gas within 24-48 hours at 44°C (HMSO, 1982). Two methods are commonly used to recover coliforms, the most probable number (MPN) and the membrane filtration (MF) technique. The technique employed to quantify these bacteria in the laboratory was the membrane filtration method. This method was chosen because of its proven reliability and reproducibility. In addition, it is less time consuming than the MPN method, and results are produced within 24 hours. The medium used was MF-Broth. This medium was purchased from commercial sources (Difco, BBL).

All sewage dilutions and filtrations were made within 6 hours of sample collection. Three different dilutions were carried out on each sample and the results were averaged. If samples could not be analysed the same day as collection, dilutions were
made, refrigerated and then analysed within 24 hours. All routine samples were analysed in triplicate. Samples were incubated at 44.5°C for 14-18 hours as longer periods resulted in yellowing of the colonies.

4.3 Chemical Analyses

4.3.1 Ammonia (NH₃-N)
The Model AN-10 CHEMet test kit and the Ammonia N(NH₃) SpectroKit used to determine ammonia-nitrogen in the sewage and pond effluent samples both utilise Nessler's reagent. The addition of tartrate (Rochelle salt) solution prevented turbidity due to calcium and/or magnesium salts from interfering with the colour development. Distillation of samples was not done due to the lack of the appropriate glassware, therefore all measurements were by direct Nesslerisation.

Measurement of ammonia concentrations of reacted samples was carried out with standard colour comparator ampoules and at times with the Bausch and Lomb Mini 20 spectrophotometer at 425 nm.

Samples were normally analysed on the day of collection. Analysis was performed only on unfiltered, diluted samples.

4.3.2 5-day Biochemical Oxygen Demand (BOD)
BOD test is defined as: "a biochemical test measuring the amount of organic matter oxidized as a result of the activities of aerobic bacteria under prescribed conditions (5 days, 20°C)" (Klein, 1959). The kinetic characteristics and the performance efficiency of wastewater treatment plants are generally assessed through the BOD removal.

In the first 2 months of monitoring after the CI pond system was commissioned in Feb88, the HACH Manometric BOD system was used to measure BOD. However, this method was soon discontinued due to increasing problems with the equipment. The azide modification (to eliminate the influence of nitrifying bacteria) of the Winkler method was then utilised for determining initial and final DO concentrations (APHA 1985) during the first 3 years of the monitoring programme. Dilution water was made up using deionised water as distilled water from the still (Corning, Megapure) had an oxygen demand that was unacceptable. Influent and pond samples were analysed within 3-6 hours of collection.
Because the BOD was used as a measurement of biodegradable compounds present, nitrification inhibitor, ATU (allyl thiourea) was added routinely during analysis of all pond effluents. This prevented nitrification so that only the carbonaceous BOD was measured (Mara, 1976). ATU was not added to the influent and septage samples. Filtered (using 2 Whatman GF/D 4.7 cm, 2.7 μm pore size) samples were included in the monitoring programme in August 1989, previously only unfiltered samples were analysed.

During the 12 month period, Jul90-Jul91, the BOD test on the influent and some pond samples gave unexpected results. These results did not correlate with COD results as expected. The BOD results appeared to be suppressed. Klein (1959) reported that the BOD of tidal waters and estuaries consisting of fresh and sea water is greatly reduced by the salinity of the water. Also, Gotaas (1949) conducted an extensive experimental study on the effect of seawater on BOD results from sewage. He reported that seawater did not significantly affect the magnitude of the BOD when less than 25% seawater was present. He also observed that seawater does not significantly affect the first stage (up to 13 days incubation) of the biochemical oxidation of sewage.

More recently, Davies et al (1978) conducted studies that showed erratic results in BOD tests on highly saline wastewaters (NaCl = 30000 mg/l) when a non-saline dilution water was used. They recommended using a saline dilution water when performing BODs on saline wastewaters. In the case of the CI pond system, the salinity was rising during the period mentioned earlier. It may be assumed that the reason for the low BOD is due not only to the resulting dilution (from saline groundwater infiltration) of the ‘true’ sewage, but also the bacteriological population in the sewage are being distressed in the presence of salt water. Another point is that during the test itself, bacteria in the sample are placed in a dilution water that is not saline, therefore creating a shock situation for microbes that are salt tolerant. It is the opinion of this author that this problem warrants more research investigation.

4.3.2.1 The Selection of BOD methods
In May 1991 a YSI BOD bottle probe for DO determinations was purchased. Parallel tests were run with the titration method 13 times on each of the weekly samples (13 weeks of samples). Comparison of results is shown in Figs 4.4-4.10.
Fig 4.4 Unfiltered and filtered BOD₅ titration and probe results for incoming composite sewage (ISC) samples, May-June, 1991.

Fig 4.5 Unfiltered and filtered BOD-5 titration and probe results for incoming grab sewage (ISG) samples, May-June, 1991.
Fig 4.6 Unfiltered and filtered BOD-5 titration and probe results for Pond 1.1 effluent samples, May-June, 1991.

Fig 4.7 Unfiltered and filtered BOD-5 titration and probe results for Pond 1.2 effluent samples, May-June, 1991.
Fig 4.8 Unfiltered and filtered BOD-5 titration and probe results for Pond 2.1 effluent samples, May-June, 1991.

Fig 4.9 Unfiltered and filtered BOD-5 titration and probe results for Pond 2.2 effluent samples, May-June, 1991.
Comparing the results for BOD titration and the probe there is generally good agreement in almost all samples of filtered raw sewage (both grab and composite) samples. The agreement between titration and probe is also good between grab samples of raw sewage which are unfiltered, and most composite raw samples.

Unusual results occurred with the incoming sewage samples (both grab and composite samples) in week 4. It is suspected that the samples were switched, however this was impossible to prove. The results for pond 1.1 in weeks 7 and 8 showed a similar discrepancy between the unfiltered and filtered samples as did pond 1.2 results in week 8. Septage samples also showed excellent agreement. During week 9-11 there were problems with the dilution water container. It was in need of cleaning as the uptake of oxygen in the blank was between 0.3 mg/l and 0.6 mg/l. After the container was properly cleaned, and put back in service during the 12th week, the oxygen uptake was reduced to between 0.0 mg/l and 0.2 mg/l.
A comparison of methods was also undertaken using standard HACH BOD ampoules which contain 150 mg/l glucose and 150 mg/l glutamic acid. The dilution water was seeded with Polyseed which is a commercial seed approved by the United States Environmental Protection Agency (US EPA). After this extensive comparative study, the probe method was chosen to replace the titration method. This increased the time available so that the laboratory could engage in other analyses.

The main advantages of using the probe over the Winkler method are: less chances of errors in measurement; speed of measurement; only a single bottle required; reduced costs for reagents; and less susceptibility to interfering substances. The opportunity for errors to occur are greatest in the Winkler method as this procedure involves at least 10 steps before a dissolved oxygen concentration can be calculated. In contrast the membrane electrode method involves only 2 steps.

In 23 Irish water pollution laboratories Fitzmaurice and Gray (1987) carried out an interlaboratory precision test for BODs. The repeatability and reproducibility of the BOD test at three concentration levels was investigated. The results showed that the membrane electrode is more precise than the Winkler method at each concentration level. The poor performance of the Winkler method was attributed to the influence of random errors caused by poor quantitative techniques. Studies conducted by the US Environmental Protection Agency (US EPA, 1978) showed comparable precision of results for the membrane electrode method.

Problems and inadequacies that are intrinsic to the traditional BOD test were pointed out by Davies et al (1977), Logan and Wangenseller (1993), and Fadini and Jardim (in press):

- wide variation in results
- susceptible to toxic compounds
- seeding may be necessary
- labour intensive
- time consuming
- conditions in lab do not mimic 'real environmental' conditions
- operational difficulties may arise during incubation resulting in unacceptable results.
The above authors suggested that the chemical oxygen demand (COD) test may be more appropriate for assessing plant performance. Sawyer and McCarty (1978) pointed out that the disadvantage of using COD is that there is no differentiation between biologically available organic matter and chemically oxidisable matter.

4.3.3 Chemical Oxygen Demand (COD)

The COD is a measurement of the amount of oxygen needed for chemical oxidation of matter, providing an estimate of the organic matter concentration in wastewater. The dichromate reflux method was utilised for the determination of COD. The HACH micro COD digestion apparatus was used. This method is simple to use as COD vials are premixed with all reagents. Filtered samples (using 2 Whatman GF/D 4.7 cm, 2.7 µm pore size) were included in the monitoring programme in August 1989, previously only unfiltered samples were analysed. The sample is added and then digested with the HACH heat reactor. As the salinity of the sewage and pond samples increased it became necessary to dilute by 6 in order to ensure that the chloride levels would not interfere with the analysis. However as the salinity decreased in 1993 and the COD measurements became lower, difficulties were experienced in getting acceptable results. In order to avoid dilution of samples with low COD, the method was adjusted in late 1993 with the addition of mercuric sulphate to avoid interference due to chlorides.

After digestion, the COD value can be determined by two methods; titrimetric and spectrophotometric. According to HACH the accuracy, reproducibility and limit of detection are similar regardless of the method of measurement used (Gibbs, 1987). The WAC laboratory determined COD values by both methods for 2.5 years (Nov89-Jul92). Standard solutions were analysed with each run. It was shown that the spectrophotometric method, using the HACH DR2000 spectrophotometer, gave reasonable and reproducible results with the standards used.

The titrimetric method (Fig 4.11) often showed percentage recoveries of more than 120%. After reviewing the results on quality control charts the author made the decision to accept the COD results from the spectrophotometer except in cases when the spectrophotometer was not available or the results were questionable. Although the DR2000 spectrophotometer gave good results, when it functioned, many problems were experienced with the instrument. The HACH company replaced the instrument twice in 3 years due to problems with unstable readings. In view of the problems experienced and the length of time the equipment was out for repairs, this author would hesitate to recommend that the HACH DR2000 be used, but would recommend
another brand such as the Bausch and Lomb 20 spectrophotometers which have proven to be sturdy and consistently reliable.

Fig 4.11 Laboratory staff member titrating sewage samples for COD.
4.3.4 Chloride (Cl)
The chloride concentrations of the samples were determined by the argentometric method as described in Standard Methods (APHA, 1985). Samples were analysed within 1-2 days.

4.3.5 Dissolved Oxygen (DO)
Dissolved oxygen measurements were carried out in Jan-Jul89, with the Winkler titration method. In May, 1990, a YSI 5739 DO probe and HACH 16046 meter was purchased. The probe is a polarographic electrode which uses a Clark-type membrane and is suitable for measurements in polluted or highly coloured, turbid wastewaters. This probe was utilised for all field DO measurements. In field use, this probe gave comparable results with the Winkler method provided that the membrane was changed at least every two weeks.

4.3.6 Hydrogen Sulphide (H₂S)
A Type S CHEMet Kit was used for the determination of hydrogen sulphide in sewage and pond effluent samples. This kit uses the "methylene blue" method which is described in Standard Methods (APHA, 1985). Measurements were carried out by colour comparison with commercial standards and at times with a Bausch and Lomb Mini 20 spectrophotometer at 670 nm.

As the levels of sulphide in the samples was often above the range of the colour standards provided in the kit, it was necessary to dilute. Samples were diluted to obtain readings within the middle range of the standards. All samples were analysed as soon as possible after collection and normally within 1 hour. It was not possible to carry out the test on-site due to time constraints.

4.3.7 Sulphate (SO₄)
A Milton Roy SpectroKit was used for the determination of sulphate. This kit employs the turbidimetric (using barium chloride) method described in Standard Methods (APHA, 1985). Each set of analyses was carried out alongside a known, commercial sulphate standard. The percentage recovery of the standard was 100-120%.

Samples were analysed on the day of collection or stored at 4°C for no longer than 24 hours. Analysis was carried out on unfiltered samples since dilutions were necessary because of the high levels of sulphate in the sewage and pond effluents. Measurement
of sulphate concentrations in reacted samples was carried out with a Bausch and Lomb Mini 20 spectrophotometer at 420 nm.

4.4 Physico-chemical Analyses

4.4.1 Physical Appearance of Ponds
From near to the start up of pond operation the appearance of ponds was noted for subsequent comparison with chemical and biological data, and as an aid to the interpretation in gross biochemical changes in the ponds. A record sheet was developed which also included weather conditions so that these too could be integrated into subsequent analysis of changes in the ponds.

4.4.2 Colour
The colour of the samples was determined by the human eye and recorded as thus. As subjective as it may seem, this method of colour identification in the ponds proved to be repeatable and recognised in literature as a way of predicting behaviour (Brockett, 1977; WHO, 1987b; and Gray, 1989).

4.4.3 Electrical Conductivity (EC)
These measurements were carried out using a WTW LF91 conductivity meter and an epoxy probe. This meter proved to be considerably more reliable than the HACH total dissolved solids (TDS) and conductivity meter which was used briefly in 1989. The HACH meter was prone to losing calibration every week, however the WTW needed calibration only once a month. Additionally, the HACH meter has a fixed ratio of 0.5 for the TDS to the EC. Calibration was carried out using standard solutions containing 10000 mg/l of NaCl. EC readings were normally taken on the day samples were collected.

4.4.4 Odour
Odour characterisation of the samples was limited by the nasal sensitivity of the human body to; very offensive; offensive; slightly offensive; not offensive; and herbal. It is recognised by the author that this is a highly subjective method of odour identification.

4.4.5 pH
Samples for pH analysis were analysed using a Fisher Accumet Ion Analyzer with a Fisher combination pH electrode and also with an Orion meter and electrode. These meters and electrodes were regularly calibrated using commercial standards and gave
satisfactory service. Analysis of samples normally occurred within 2-5 hours after collection. On-site measurements were not possible as the meters were not portable.

4.4.6 Sludge
Measurements of sludge accumulation were carried out using the 'white towel' method (Fig 4.12) recommended and described by Pearson et al (1987a).

4.4.7 Temperature
Temperature readings were taken immediately as samples were collected. A mercury-filled field thermometer with a range of -10°C to 110°C and 0.5°C graduations was used.
Maximum and minimum temperature measurements were measured weekly using max/min thermometers suspended mid-depth in the middle of each pond for the period Apr90-Jan92.

4.4.8 Total Suspended Solids (TSS)
Suspended solids were determined as outlined in Standard Methods (APHA, 1985) using Whatman 954AH 2.1 cm filters (1.5 μm pore size), Gooch crucibles and drying at 103-105°C for 1 hour.

4.4.9 Total Volatile (TVSS) and Fixed Suspended Solids (TFSS)
The dried residue in the Gooch crucible from the TSS analysis was used for determining volatile and fixed suspended solids at 550 °C in a muffle furnace.

4.5 Quality Assurance and Quality Control
It is critical that any analysis and data reported whether for publication or internal use be reliable and trustworthy. Quality assurance and quality control are two aspects of laboratory management that ensure the production of reliable data. For those reasons, these very important aspects of laboratory monitoring where given considerable attention.

4.5.1 Quality Assurance
In order to ensure that data produced were reliable, certain procedures were followed routinely, these included:

1. Training of laboratory personnel.
2. Proper cleaning and washing of glassware and apparatus.
3. Proper maintenance and calibration of instruments and equipment.
4. Standardisation of normal solutions.
5. Development of standard forms for every analytical procedure.
6. Consistent record-keeping of field conditions, analyses, and instruments.
7. Use of standard analytical methods.
8. Adequate consideration and standardisation of sampling methods and preservation.

4.5.2 Quality Control
Although the laboratory did not participate in external analytical quality control programmes, internal quality control was practiced for parameters where standard solutions were available. Percentage recovery control charts were developed with
upper and lower warning limits. The establishment of control charts and regular analysis of solutions of known standard concentrations enabled the laboratory to ensure that precision remained adequate.

4.6 Meteorological Data

Meteorological data for total rainfall, rainfall intensity and windspeed were monitored daily at the sewage treatment works, rainfall since 1989 and windspeed since 1990.

Additionally data was provided by the Civil Aviation Meteorological Station located at the Owen Roberts International Airport in Grand Cayman. This station is approximately 2 km from the treatment works. Data provided included monthly averages for windspeed, wind direction, rainfall, air temperature and relative humidity for the period 1988-1994.

Climatic information was also made available by the DoE, Research Unit (formerly the Mosquito Research and Control Unit, MRCU) which collects climatic data for the World Meteorological Organisation (WMO). This meteorological station is located in West Bay, approximately 8 km from the treatment works. The data provided included daily hours of sunshine, evaporation rates, solar radiation and relative humidity.
CHAPTER 5

5.0 ROUTINE MONITORING PHASE I MARCH 1988 TO MARCH 1990

In this chapter and the following 2 chapters the results of the routine monitoring programme are reported and discussed. For practical presentation and discussion, the monitoring periods are divided into 3 major phases. These are:

1. Monitoring Phase I Mar88-Mar90, which encompasses the start-up period, saline groundwater infiltration, baffle installation, and aerators installation.

2. Monitoring Phase II Apr90-Mar93, which shows the continuous rise in flow and salinity of the raw sewage.

3. Summary and Monitoring Apr93-Dec94, during which the sewer structures were rehabilitated and repaired resulting in partial recovery of the performance efficiency of the sewage treatment lagoons.

5.1 Commissioning of Sewage Treatment Ponds February 1988

Construction of the Cayman Islands sewerage and sewage treatment works were completed in February, 1988, and the entire system was commissioned the same month. The ponds had retained some rainwater after construction, and consequently had a healthy population of algal growth indicated by the vibrant green colour of the liquid (Fig 5.1).

The majority of major connections (hotels and condominiums) to the sewerage system was completed by September, 1988. Recirculation of the effluent from maturation pond 2.2 commenced on 30 May of the same year. The deepwells began to receive the majority of the final effluent in June. Septage was accepted at the works from 22 July, 1988.
As the final effluent was to be sold, a basic monitoring programme was developed (described in the previous chapter) and implemented in March, 1988. The main focus of the monitoring programme was aimed at determining the general performance of the system and to determine whether the final effluent met the standards required for reuse. Salinity was a very important parameter of this monitoring programme because the effluent would be used to irrigate golf-course grass which could not survive >3500 μS/cm salinity (measured as electrical conductivity, EC). Additionally, the other concern was the intactness of the sewerage system as a high percentage of the sewers and all of the interceptors are situated below the saline groundwater table (Fig 5.5 and 5.3).

Fig 5.1 Facultative pond 1.1 prior to being commissioned showing the collected rainwater and green colour indicating a vibrant algal population.
Fig 5.2 High groundwater table necessitate professional divers to assist in laying sewer mains and interceptors along West Bay Road.

Fig 5.3 Professional diver assisting in the positioning of pre-cast sulphate resistant concrete pumping station rings.
5.2 Saline Groundwater Infiltration

In June of the start-up year major groundwater infiltration into the sewer lines was discovered through the monitoring programme. Electrical conductivity increased from <2000 μS/cm to almost 5000 μS/cm (Fig 5.4). The infiltration point was repaired and subsequently the salinity was reduced to an acceptable level for reuse in Jul88.

Fig 5.4 Average daily incoming sewage flow and electrical conductivity for the period Mar88-Mar90 (Phase I).

However, the benefits of repairs were temporary and in Sep88, the Cayman Islands were affected by hurricane Gilbert. This resulted in the interceptors within the sewerage drainage area being flooded by rising sea-levels causing the lagoons to be inundated. Additionally, there was significant structural damage to the sewage treatment site (Fig 5.5).
Salinity became a concern, because in addition to the hurricane flooding, 2 major hotels that utilise seawater for toilet flushing were connected to the system during Sep88.

The incoming sewage flow during the first year of operation (Fig 5.4) exhibited a steady increase. This was as expected since more connections were being made over that period (Sep88-Jun89). After the hurricane, the EC levels of the incoming sewage remained fairly stable at an average of 5790 μS/cm (Fig 5.4). The incoming flow for the later part of the same period also remained stable, averaging 1385 m³/day. For
almost 1 year after the hurricane, the sewer system seemed to have stabilised as there were no significant sustained increases in salinity during that period. Although the number of connections increased and even with the onset of the tourist season in Dec88, the inflow and EC of the inflow to the system showed only slight fluctuations. This indicated that the system was operating with satisfactory stability (with the exception of odour problems).

The process of deterioration of the sewerage system structures did not begin to accelerate until Oct89 when it was observed that the salinity of the raw sewage was increasing again with corresponding increases in flow (Fig 5.4 and 5.7). The average daily flow increased from 1547 m³/day in Sep89 to 1764 m³/day in Oct89. By the end of monitoring phase 1 in Mar90, the average daily flow of sewage to the works had reached 2634 m³/day, an increase of 70% from the levels in Sep89. The average EC of the raw sewage was also rising during this period and by the end of Mar90, was 35% higher (from 7730 to 10498 μS/cm) than it was in Sep89.

In Fig 5.6 the incoming flow to the works and EC for the period Mar88-Sep89 were plotted against each other. The linear regression curve did not show a highly significant correlation ($R^2 = 0.63$). This suggests that the rise in flow at that time was not attributable to saline groundwater infiltration alone. The correlation between salinity and flow enables the testing of 3 hypotheses:

1. Increasing infiltration from groundwater is the main cause of increasing flows. If this is the case then there should be an 'equivalent' rise in conductivity, therefore a high ‘$R$’ (Fig 5.7).

2. Increasing connections are the main cause of increasing flow. If this is the case then there should not be a corresponding rise in EC (poor correlation), unless most hotels are discharging saline sewage.

3. Both new connections and infiltration are contributing to increasing flow. If this is the case, flow and conductivity should be at an intermediate level of correlation (Fig 5.6).

An intensive investigation of salinity levels in the sewage along the sewerage system, in addition to observations on the physical structures of the sewers, lift stations and manholes, indicated that there were a number of leaks and cracks developing within the
structures. The rising salinity and flow levels were due mainly to inflow of groundwater through the structural failures discovered

![Graph 1: Correlation between monthly average daily sewage flow and average electrical conductivity for period Mar88-Sep89.](image)

![Graph 2: Correlation between monthly average daily sewage flow and average electrical conductivity for period Oct89-Mar90.](image)

In Fig 5.7, where the same parameters are plotted for the period Oct89-Mar90, a significant correlation of $R^2 = 0.81$ was obtained, demonstrating that the rising flow was closely linked with increased saline groundwater infiltration into the sewers.

The average nominal retention of both facultative ponds was 24 days, the maturation ponds 2.1 and 2.2 was 8 and 7 days respectively. Accordingly for the entire treatment system the mean nominal retention time, based on pond design volume and flow for the period Jul88-Mar90, was approximately 40 days.

### 5.3 Organic Loading and BOD Removal

Because of the additional 'non-sewage' water entering the system, the organic load and suspended solids were diluted to levels lower than that of the design as demonstrated by the results of the monitoring programme.

During the only period of low salinity in the raw sewage (from Mar88 to Aug88), the BODuf (unfiltered) of the influent ranged from 127.5 mg/l to 328.3 mg/l with a mean of 224 mg/l (Fig 5.8 and Fig 5.10). The BODuf of the final effluent, during the same period, ranged from 7.3 mg/l to 10.1 mg/l with a mean of 8.5 mg/l (Fig 5.9).
From Fig 5.8, the BODuf removal in the facultative ponds averaged 81% for the period Mar88-Aug88 (before salinity problems). During the period when the salinity of the incoming sewage stabilised (Sep88-Sep89), the BODuf removal in the facultative ponds decreased by 10% to an average of 71%. From Oct89 to Mar90, when the EC of the incoming sewage was gradually increasing, the BODuf removal improved slightly to 75%. This improvement coincided with the installation of the aerators in the facultative ponds (see Section 5.5) and with reducing raw sewage BOD due to dilution by saline water.

In the first 6 months of operation, overall BOD removal efficiency was at its highest (96%) compared to the following months (Fig 5.9) due to underload and dilution into rain water present in the ponds prior to commission. Both facultative ponds, 1.1 and 1.2 performed well, with individual removal efficiencies of 76 and 85%, respectively.
BODuf removal in the first maturation pond 2.1 averaged 45% over the monitoring period Mar88-Aug88, then in the period Sep88-Sep89 efficiency dramatically decreased to an average of 16%. When the salinity levels began to gradually rise, BODuf removal efficiency did not exhibit a dramatic change, averaging 17%. In maturation pond 2.2, the BOD removals were lower than in maturation pond 2.1 for the period of low saline sewage (Mar88-Aug88), averaging 28%. The removal efficiency of the final maturation pond for the Sep88-Sep89 period decreased further to 15%. Despite the rising EC in Oct89-Mar90, mean BODuf removal was 24%. During the operation of the WSP for the entire monitoring phase I, the overall BOD removal efficiency was 87%.

In Jul89 analysis of the filtered BOD(f) became a part of the routine monitoring programme. The subsequent results showed that the soluble BODf contributed 42% of
the total BODuf in the raw sewage. The level was similar in the facultative ponds at 41%, while in the maturation ponds the soluble BOD was 36% of the total BOD.

The BODuf surface loading on the facultative ponds for the monitoring period Mar88-Mar90 averaged 238 kg/ha day and ranged from 54 to 346 kg/ha day. The loading was considerably lower than that of the design of 420 kg/ha day.

The effect of saline groundwater intrusion was generally one of dilution. Fig 5.10 shows the flow pattern of the incoming sewage and the concentration of BODuf in the raw sewage for the phase I monitoring period.

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**Fig 5.10** Monthly average BODuf of incoming raw sewage and the average daily flow to the sewage treatment works during the period Mar88-Mar90.

A linear regression of raw sewage BODuf data and daily average flow for the period Mar88-Sep89 did not indicate any significant correlation (Fig 5.11). This is probably
because the start-up period incorporated a number of unusual events including a sewer break and repair, the hurricane, and the initial rainwater.

![Graph](image1)

![Graph](image2)

Fig 5.11 Correlation between monthly average raw sewage BODuf and monthly average of daily inflow during the period Mar88-Sep89.

Fig 5.12 Correlation between monthly average raw sewage BODuf and monthly average of daily inflow during the period Oct89-Mar90.

However, the plot of raw sewage BODuf data and daily average flow for the subsequent period Oct89-Mar90, gave a good correlation of $R^2 = 0.82$ (Fig 5.11). This confirms that the increasing flow is consistently and increasingly diluting the organic strength of the sewage and is therefore not purely domestic wastewater.

5.3.1 COD Removal

Analysis for COD as an indicator of the WSP system performance was carried out on a few samples in 1988, however it was not until mid 1989 that CODuf and CODf became a regular part of the monitoring programme.

The incoming sewage CODuf values varied from 361 to 510 mg/l and those of the final effluent from 120 to 241 mg/l during monitoring Phase I. The CODuf removal in the facultative ponds averaged 47.5%, while the maturation ponds 2.1 and 2.2 averaged 54% and 1% respectively (Table 5.1). During this monitoring period the mean CODuf surface loading on the facultative ponds was 428 kg/ha day and was 217.5 kg/ha day and 230 kg/ha day on the maturation ponds 2.1 and 2.2, respectively.
Table 5.1  Average CODuf and CODf in sewage and sewage treatment ponds, and percentage removals throughout the treatment system (Mar88-Mar90).

<table>
<thead>
<tr>
<th></th>
<th>In Sew</th>
<th>PD 1.1 avg.</th>
<th>% rem.</th>
<th>PD 1.2 avg.</th>
<th>% rem.</th>
<th>PD 2.1 avg.</th>
<th>% rem.</th>
<th>PD 2.2 avg.</th>
<th>% rem.</th>
<th>Overall % rem.</th>
</tr>
</thead>
<tbody>
<tr>
<td>CODuf</td>
<td>421</td>
<td>222</td>
<td>47</td>
<td>220</td>
<td>48</td>
<td>192</td>
<td>54</td>
<td>191</td>
<td>4</td>
<td>55</td>
</tr>
<tr>
<td>CODf</td>
<td>325</td>
<td>160</td>
<td>51</td>
<td>146</td>
<td>55</td>
<td>109</td>
<td>67</td>
<td>113</td>
<td>4</td>
<td>65</td>
</tr>
</tbody>
</table>

The CODf of the influent showed little significant variation during this monitoring phase ranging from 263 to 388 mg/l and averaged 325.4 mg/l representing 77% of the total COD. The CODf of the final effluent varied from 89 to 149 mg/l and averaged 113 mg/l. CODf removal efficiency in the facultative ponds was slightly higher than that of CODuf at 51-55% (Table 5.1). Removal in maturation ponds 2.1 and 2.2 was 67% and 4%, respectively. During this period the mean CODf surface loading on the facultative ponds was 382 kg/ha day and was 168.5 kg/ha day and 113 kg/ha day on the maturation ponds 2.1 and 2.2, respectively. Overall CODuf and CODf removal was 55% and 65% respectively for Phase I monitoring period.

The COD compared to the BOD is invariably expected to be higher since it includes substances that are chemically oxidised as well as biologically oxidised (Gray, 1989). According to Gray (1989) the COD:BOD relationship varies from 1.25 to 2.50 depending on the type of waste being analysed and if the ratio is found to be >4.1 it is an indication of the possible presence of toxic compounds.

The ratio of CODuf:BODuf in the incoming sewage during the monitoring phase I, averaged 2.7, (Appendix I). This ratio is slightly higher than that suggested by Gray (1989). However the relationship of CODf:BODf in the raw sewage averaged 5.0. This indicates that there are more chemically oxidisable compounds in the raw sewage than are normally found in domestic wastewater.

The ratio increases with each stage of biological treatment as biodegradable matter is consumed but the non-biodegradable organics remain and are oxidised in the COD test (Gray, 1989). This was the case in the facultative and maturation ponds where the average ratio of CODuf:BODuf was 6.4 and 7.9, respectively.
5.4 Faecal Coliform Removal Efficiency

From Fig 5.13, during the first 6 month period of operation the maturation ponds showed excellent faecal coliform removals such that the design target of <100 faecal coliforms cfu/100 ml was met from Jun88-Sep88.

In the second period (Sep88-Sep89), the primary ponds have a high removal efficiency, while the secondary ponds show a very substantial 1 log reduction in performance compared to the first 6 month phase (Table 5.2). This deterioration in the performance of the maturation ponds occurred as increasing quantities of saline water entered the system in Sep88.

The % removal efficiency of the facultative ponds in removing faecal coliform indicator bacteria improved further in the 3rd period (Oct89-Mar90) of this monitoring phase.
This may have been influenced by the installation of the aerators in Oct89 which led to the development of algae in the ponds. The development of toxic conditions associated with anaerobic conditions, which would hinder algal development (and the development of predators), were presumably restrained by the mechanical aeration. Photooxidation has been found to be an important factor in faecal coliform removal mechanisms (Curtis, 1990) but much less work has been done on predators in lagoons (Benson-Evans, 1975). Algal activity would normally be expected to have its major effect in maturation ponds. This is not generally the case here.

5.4.1 Effect of Salinity on Faecal Coliform Removal Efficiency
During this the first year of operation, the WSP system experienced a number of changes. As discussed previously in this chapter, salinity was only one of them. It is therefore very difficult to attribute the changes in behaviour observed to any specific factor. More likely it is a combination of factors that influence these changes. The influence of salinity on the sewage treatment system was investigated as the raw sewage underwent several changes in saline concentration. Throughout the entire period the plant operated within the design flows. Table 5.2, shows the % removals of faecal coliforms in each pond of the lagoon system during the different salinity periods.

Table 5.2 shows that faecal coliform removal efficiencies in the facultative ponds were not adversely affected by the increased salinity and flow. In fact, in spite of dilution the raw sewage faecal coliform counts rose steadily throughout phase 1 (Fig 5.13 - IS Grab) but the level at the outlets of pond 1.1 and 1.2 remained steady at $10^5$ for 18 months. These facultative ponds thus improved in the removal of faecal coliforms. However both maturation ponds, demonstrated markedly reduced removal efficiencies, especially the final maturation pond 2.2, which had an average of only 76% removal efficiency. This represents a reduction in performance by approximately 20% from 95% prior to the highly saline sewage entering the system.
Table 5.2 Percentage faecal coliform removal and electrical conductivity for different salinity periods.

<table>
<thead>
<tr>
<th>PERCENTAGE FAECAL COLIFORM REMOVALS &amp; CONDUCTIVITY (EC) μS/cm</th>
<th>Mar88-Aug88</th>
<th>Sep88-Sep89</th>
<th>Oct89-Mar90</th>
</tr>
</thead>
<tbody>
<tr>
<td>PD1.1 Facultative Pond</td>
<td>98.96</td>
<td>99.17</td>
<td>99.98</td>
</tr>
<tr>
<td>EC</td>
<td>1618</td>
<td>5689</td>
<td>9352</td>
</tr>
<tr>
<td>PD1.2 Facultative Pond</td>
<td>99.52</td>
<td>99.48</td>
<td>99.98</td>
</tr>
<tr>
<td>EC</td>
<td>1856</td>
<td>5639</td>
<td>9278</td>
</tr>
<tr>
<td>PD2.1 Maturation Pond</td>
<td>92.28</td>
<td>83.55</td>
<td>88.34</td>
</tr>
<tr>
<td>EC</td>
<td>1680</td>
<td>5686</td>
<td>9443</td>
</tr>
<tr>
<td>PD2.2 Maturation Pond</td>
<td>95.02</td>
<td>86.91</td>
<td>76.44</td>
</tr>
<tr>
<td>EC</td>
<td>1684</td>
<td>5684</td>
<td>9243</td>
</tr>
<tr>
<td>Overall removal</td>
<td>99.998</td>
<td>99.98</td>
<td>99.999</td>
</tr>
<tr>
<td>Raw sewage EC</td>
<td>2205</td>
<td>5790</td>
<td>9544</td>
</tr>
</tbody>
</table>

The net result in the WSP system is one of improved removals as evidenced by the ‘overall removal’ efficiencies shown in Table 5.2. From the above it may be concluded that the facultative ponds are able to acclimatise more readily to rises in salinity (<10000 μS/cm) than the maturation ponds which seem to be more sensitive to salinity or associated factors.

5.4.2 Effect of pH on Faecal Coliform Removal Efficiency

As the average pH of the ponds in the system showed dramatic changes during the period that the EC was increasing in the raw sewage, the data were further investigated. Table 5.3, shows the % removals of faecal coliforms in each pond of the lagoon system during the different salinity periods.
Table 5.3 Percentage faecal coliform removal and average pH in WSP system for different salinity periods.

<table>
<thead>
<tr>
<th>PERCENTAGE FAECAL COLIFORM REMOVALS &amp; pH units</th>
<th>Mar88-Aug88</th>
<th>Sep88-Sep89</th>
<th>Oct89-Mar90</th>
</tr>
</thead>
<tbody>
<tr>
<td>PD1.1 Facultative Pond</td>
<td>98.96</td>
<td>99.17</td>
<td>99.98</td>
</tr>
<tr>
<td>pH</td>
<td>9.1</td>
<td>8.0</td>
<td>7.6</td>
</tr>
<tr>
<td>PD1.2 Facultative Pond</td>
<td>99.52</td>
<td>99.48</td>
<td>99.98</td>
</tr>
<tr>
<td>pH</td>
<td>9.6</td>
<td>8.0</td>
<td>7.5</td>
</tr>
<tr>
<td>PD2.1 Maturation Pond</td>
<td>92.28</td>
<td>83.55</td>
<td>88.34</td>
</tr>
<tr>
<td>pH</td>
<td>9.8</td>
<td>8.0</td>
<td>7.6</td>
</tr>
<tr>
<td>PD2.2 Maturation Pond</td>
<td>95.02</td>
<td>86.91</td>
<td>76.44</td>
</tr>
<tr>
<td>pH</td>
<td>10.0</td>
<td>8.1</td>
<td>7.8</td>
</tr>
<tr>
<td>Overall removal</td>
<td>99.998</td>
<td>99.98</td>
<td>99.999</td>
</tr>
<tr>
<td>Raw sewage pH</td>
<td>7.5</td>
<td>7.6</td>
<td>7.4</td>
</tr>
</tbody>
</table>

In order to determine whether there was any detectable effect on the faecal coliform removal efficiency in the WSP related to the changing pH calculations to determine the correlation coefficient ($R^2$) were carried out on the pH and faecal coliform % removal data. These results are shown in Table 5.4:

Table 5.4 Correlation coefficients for percentage faecal coliform removals and average pH in WSP system for different salinity periods.

<table>
<thead>
<tr>
<th>Correlations ($R^2$) between faecal coliform % removal and pH</th>
<th>PD 1.1</th>
<th>PD 1.2</th>
<th>PD 2.1</th>
<th>PD 2.2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mar88-Aug88</td>
<td>0.76</td>
<td>0.86</td>
<td>0.84</td>
<td>n/d</td>
</tr>
<tr>
<td>Sep88-Sep89</td>
<td>0.08</td>
<td>0.05</td>
<td>0.16</td>
<td>0.00</td>
</tr>
<tr>
<td>Oct89-Mar90</td>
<td>0.05</td>
<td>0.01</td>
<td>0.25</td>
<td>0.35</td>
</tr>
</tbody>
</table>

*note: n/d = no data, pond being filled*

It is notable that the pH versus % FC removal correlations are significant only during the start-up period. This raises many questions, for example; is $H_2S$ production
affecting pH or vice versa? \( \text{H}_2\text{S} \) produced in the facultative ponds is transferred to maturation ponds where algal production is inhibited. Consequently the high pH associated with photosynthesis in maturation ponds is not observed.

5.5 Effect of Baffles Installed in Maturation Ponds

Information on the dimensions and location of the baffles in the maturation ponds was given in Chapter 3.

Baffles were installed in Mar89 in the maturation ponds in order to improve faecal coliform removal efficiencies through improved retention time. Most discussion on sewage treatment in lagoons in relation to baffles is concerned with suspended biomass and its influence on BOD removal or decomposition (Oswald and Gotaas, 1955; Gloyna, 1971; Mara, 1976; and Middlebrooks et al, 1982). Muttamara and Puerpaiboon (in press) reported that baffles tend to act as a habitat for growth of attached-growth bacteria and algae thereby increasing pond biomass.

Some experimental studies have shown that ponds with baffles installed have better removal efficiencies than those without (Kilani and Ogunrombi, 1984). These authors attributed the improved performance observed to a reduction in the dispersion number, that is the hydraulic efficiency - nearer to plug flow (Chapter 10). However, Reynolds (1975) found no significant improvement in performance with the installation of baffles. Some researchers have postulated the improved performance of ponds usually observed with baffle installation may be due to the biofilm that attaches to the baffle surfaces (Shin and Polprasert, 1987; and Baskaran et al, 1992). Similarly, as in rivers and streams, (Gantzer et al, 1988), Polprasert and Agarwalla (in press) suggested that in facultative ponds as much as 46-49% BOD removal may be attributed to the biofilm biomass attached to the pond sides, bottom and baffle curtains!

In the Cayman Islands system because the installation of baffles coincided with increasing saline groundwater intrusion, it is difficult to identify the direct effect baffle installation had on the performance of the 2 maturation ponds. The general performance of the maturation ponds, in terms of faecal coliform bacteria and BODuf removal, before and after the installation of baffles is shown in Table 5.5.
Table 5.5 BOD and faecal coliform removal percentages before and after baffle installation in maturation ponds.

<table>
<thead>
<tr>
<th>PERCENTAGE REMOVALS</th>
<th>Before baffles in maturation ponds</th>
<th>After baffles in maturation ponds</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(Nov88-Mar89)</td>
<td>(Apr89-Jul89)</td>
</tr>
<tr>
<td></td>
<td>avg daily flow 1469 m$^3$/day</td>
<td>avg daily flow 1393 m$^3$/day</td>
</tr>
<tr>
<td>Mat Pd 2.1</td>
<td>Mat Pd 2.2</td>
<td>WSP system</td>
</tr>
<tr>
<td>Mat Pd 2.1</td>
<td>Mat Pd 2.2</td>
<td>WSP system</td>
</tr>
<tr>
<td>BOD$_{uf}$ mg/l</td>
<td>23</td>
<td>-4</td>
</tr>
<tr>
<td>Faecal coliform cfu/100</td>
<td>72.91</td>
<td>65.60</td>
</tr>
<tr>
<td>ml</td>
<td>9</td>
<td>95.87</td>
</tr>
<tr>
<td></td>
<td>79</td>
<td>99.9689</td>
</tr>
<tr>
<td></td>
<td>77</td>
<td>99.9959</td>
</tr>
</tbody>
</table>

There was no improvement in terms of BOD$_{uf}$ removal in the first maturation pond, in fact there was an overall average increase in BOD$_{uf}$.

The opposite effect was seen in the BOD$_{uf}$ removals in the final maturation pond 2.2, where there was a significant increase from 83.33% in Nov88-Mar89 to 95.87% in the 4 month period after the baffles were installed (Apr89-Jul89). This indicates that the BOD$_{uf}$ removal in the final maturation pond was improved but there was no improvement in maturation pond 2.1. It may be hypothesised that the reason for this is the combined effect of the baffles and wind driven shortcircuiting. Wind effects will be further discussed in Chapter 10.

The evidence shows that before the baffles were installed, maturation pond 2.2 was already more effective (83%) than maturation pond 2.1 (73%) in faecal coliform removal. The baffles appear to have enhanced both positive and negative effects. One probable explanation is the location of windows in the baffles (Fig 3.10, pg 54) causing a reduction in surface streaming/shortcircuiting down the pond and increasing shortcircuiting in the direction of the wind.

For the 4 months prior to baffle installation, the faecal coliform removal in maturation pond 2.1 was 72.91%. However after installation, for the period Apr-Jul89, the removal efficiency decreased to 65.60%. In the final maturation pond (2.2), the effect of the baffles is clearly seen in the faecal coliform removals for the same periods. The faecal coliforms removed from this pond increased from an average of 83.3%, in the period Nov88-Mar89, to an average removal of 95.87%. Additionally the overall removal efficiency of the WSP treatment system improved from producing a final effluent showing a 3 log removal, to one with more than 4 log removal in the 4 months after the baffles were installed.
The faecal coliform data for the incoming sewage and the final effluent (before and after baffles were installed) are presented in Table 5.6.

Table 5.6  Faecal coliform bacteria densities in raw sewage and final pond effluent, and percentage removals for 4 months before and after baffles were installed in the maturation ponds.

<table>
<thead>
<tr>
<th>DATE</th>
<th>Fecal coliform density (cfu/100 ml)</th>
<th>% removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>3-Nov-88</td>
<td>1.95E+06</td>
<td>99.9348%</td>
</tr>
<tr>
<td>10-Nov-88</td>
<td>4.90E+06</td>
<td>99.9917%</td>
</tr>
<tr>
<td>17-Nov-88</td>
<td>4.27E+06</td>
<td>99.9446%</td>
</tr>
<tr>
<td>24-Nov-88</td>
<td>7.00E+05</td>
<td>99.0286%</td>
</tr>
<tr>
<td>22-Dec-88</td>
<td>2.17E+07</td>
<td>99.9676%</td>
</tr>
<tr>
<td>5-Jan-89</td>
<td>3.20E+07</td>
<td>99.9525%</td>
</tr>
<tr>
<td>12-Jan-89</td>
<td>1.47E+07</td>
<td>99.9472%</td>
</tr>
<tr>
<td>18-Jan-89</td>
<td>4.37E+07</td>
<td>99.9954%</td>
</tr>
<tr>
<td>25-Jan-89</td>
<td>4.23E+06</td>
<td>99.9186%</td>
</tr>
<tr>
<td>1-Feb-89</td>
<td>6.10E+06</td>
<td>99.8796%</td>
</tr>
<tr>
<td>9-Feb-89</td>
<td>1.66E+07</td>
<td>99.9670%</td>
</tr>
<tr>
<td>15-Feb-89</td>
<td>6.59E+07</td>
<td>99.9933%</td>
</tr>
<tr>
<td>22-Feb-89</td>
<td>9.57E+06</td>
<td>99.9425%</td>
</tr>
<tr>
<td>1-Mar-89</td>
<td>1.04E+07</td>
<td>99.9926%</td>
</tr>
<tr>
<td>8-Mar-89</td>
<td>2.18E+07</td>
<td>99.9927%</td>
</tr>
<tr>
<td>15-Mar-89</td>
<td>2.83E+07</td>
<td>99.9933%</td>
</tr>
<tr>
<td>Mean</td>
<td>1.79E+07</td>
<td>99.9689%</td>
</tr>
<tr>
<td>No.</td>
<td>16</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>DATE</th>
<th>Fecal coliform density (cfu/100 ml)</th>
<th>% removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>29-Mar-89</td>
<td>3.39E+07</td>
<td>99.9941%</td>
</tr>
<tr>
<td>5-Apr-89</td>
<td>1.40E+07</td>
<td>99.9857%</td>
</tr>
<tr>
<td>19-Apr-89</td>
<td>2.52E+06</td>
<td>99.9915%</td>
</tr>
<tr>
<td>26-Apr-89</td>
<td>2.40E+06</td>
<td>99.9375%</td>
</tr>
<tr>
<td>3-May-89</td>
<td>5.93E+06</td>
<td>99.9769%</td>
</tr>
<tr>
<td>11-May-89</td>
<td>4.33E+06</td>
<td>99.9686%</td>
</tr>
<tr>
<td>18-May-89</td>
<td>6.35E+05</td>
<td>99.7969%</td>
</tr>
<tr>
<td>24-May-89</td>
<td>1.76E+07</td>
<td>99.9973%</td>
</tr>
<tr>
<td>31-May-89</td>
<td>2.21E+06</td>
<td>99.9303%</td>
</tr>
<tr>
<td>9-Jun-89</td>
<td>4.29E+07</td>
<td>99.9996%</td>
</tr>
<tr>
<td>15-Jun-89</td>
<td>3.47E+07</td>
<td>99.9862%</td>
</tr>
<tr>
<td>21-Jun-89</td>
<td>1.22E+08</td>
<td>99.9990%</td>
</tr>
<tr>
<td>29-Jun-89</td>
<td>6.47E+07</td>
<td>99.9984%</td>
</tr>
<tr>
<td>5-Jul-89</td>
<td>5.42E+06</td>
<td>99.9833%</td>
</tr>
<tr>
<td>12-Jul-89</td>
<td>1.68E+07</td>
<td>99.9981%</td>
</tr>
<tr>
<td>19-Jul-89</td>
<td>1.12E+08</td>
<td>99.9995%</td>
</tr>
<tr>
<td>26-Jul-89</td>
<td>7.00E+06</td>
<td>99.9805%</td>
</tr>
<tr>
<td>Mean</td>
<td>2.84E+07</td>
<td>99.9559%</td>
</tr>
<tr>
<td>No.</td>
<td>17</td>
<td></td>
</tr>
</tbody>
</table>

The data above demonstrates the difficulties in producing a final effluent that is suitable for irrigation on public parks, which would also include golf courses. In the 4 month period before the baffles were installed, only on 4 occasions were the overall faecal coliform removal efficiency able to achieve more than a 4 log removal. Additionally the faecal coliform density in the final effluent was such that, for the period before the baffles, the effluent quality was not suitable for irrigation reuse for 87% of the time, i.e. >1000 faecal coliforms cfu/100 ml.

After the baffles were installed there was considerable improvement in the faecal coliform removals of the entire system. Improving from a 3 log removal to >4 logs for
10 out of 17 samples in the 4 month period following the baffles’ installation. In this period the bacteriological quality of the effluent was suitable for irrigation discharge for 63% of the samples analysed, i.e. <1000 faecal coliforms cfu/100 ml (IRCWD, 1985). This demonstrated that there was considerable improvement in effluent quality after baffles were installed in the maturation ponds.

Between the 2 periods, the ponds' liquid temperature increased from an average of 26°C in the 4 months prior to the baffles being installed, to an average of 30°C during the 4 months after they were installed. The intention of improving the faecal coliform removal was achieved with installation of the baffles although it may be argued that seasonal factors such as temperature and sunlight intensity may have influenced the faecal coliform removal.

5.6 Impact of Aerators on Facultative Ponds
By the end of Oct88, the physical observation records show that the ponds changed from a vibrant, dark green colour to a milky brown, coinciding with increased salinity. In addition to this physical change, offensive odours were produced. The odour problem was not appreciated by the residents downwind of the system who understandably, complained. In order to address the odour problem, the WAC had little alternative but to install 8 diffuse aerators in each facultative pond as described in Chapter 3 and shown in Figs 3.3 and 3.14. This was accomplished a year after the problem of odours and salinity first became an issue in the behaviour and efficiency of the system.

Physical observations during the first year of operation (Mar88-Mar89) showed that the all 4 ponds had a green appearance for about 70% of the year. The green colour observed, indicated that the ponds were operating with a healthy algal community. However, in the second year of operation and before the installation of the aerators, the predominate colour observed was brown, the ‘pinking’ of the ponds was also reported and occasionally they were described as brownish green in colour. This indicated poor algal development in the lagoons.

In Oct89 after the installation and operation of the aerators in the facultative ponds, the ponds returned to a vibrant green as shown in Fig 3.3, pg 44. Generally, WSP systems oxygenated through the use of diffuse or mechanical aerators exhibit reduced algae growth (Mara, 1976) due to turbulence and turbidity (Benjes, 1970; and Boyko and Rupke, 1970). Reduced algal activity did not occur in the Cayman Islands ponds even though for the first 3 months 4 aerators were used continuously day and night.
(Tables 3.4 and 3.5, pg 58). Moreover the ponds exhibited increased algal growth which by physical observations appeared to be a healthy green (Fig 3.3, pg 44).

The power-level range in which the aerators in the facultative lagoons operated was not sufficient to maintain all solids in suspension as would be the case if the ponds were operating in a true 'aerated lagoon' state. The aerators operated with a minimum power-level to maintain dissolved oxygen in the upper liquid phase and at the same time limit the resuspension of solids. This method of operation, therefore, did not inhibit algal growth and is in contrast to aerated facultative ponds described by (Benjes, 1970).

5.6.1 Suspended Solids (SS)
As suspended solids are one of the parameters likely to be affected by continual aeration (due to the resuspension of solids in the facultative ponds), it was expected that there may be some increase in SS in the facultative ponds' effluents. Because of this, the SS were carefully monitored. Effluent from aerated ponds generally range from 260 mg/l to 300 mg/l (Metcalf and Eddy, 1972).

The SS weekly data points for the incoming sewage and facultative ponds are plotted in Fig 5.14. Initially, during the start-up period and until Aug88, the SS of the incoming sewage averaged 141 mg/l, ranging from 47-261 mg/l. In the period Sep88-Sep89, during which the EC stabilised, the SS of the incoming sewage averaged 173 mg/l.

The SS in the facultative ponds' effluents averaged 153 mg/l, with a range of 17-374 mg/l, in the short period during the start-up when the sewage was not saline. The difference between the SS measured in the 2 facultative ponds (Fig 5.14) was negligible.
Fig 5.14 Weekly suspended solids data for raw sewage and the facultative ponds for the period Mar88-Mar90 (Phase I).

During the period Sep88-Sep89 when the salinity had stabilised, effluent from the facultative ponds averaged 127 mg/l, and ranged from 30-324 mg/l. No appreciable difference in the average SS in each of 2 facultative ponds was observed (Appendix III and IV).

Data obtained from the maturation ponds for the same period are plotted in Fig 5.15. The SS in the final maturation pond effluent averaged 50 mg/l, with a range of 18-115 mg/l, in the short period during the start-up when the sewage was not saline. After, the stabilising period Sep88-Sep89, this changed to an average of 88 mg/l (Fig 5.15), ranging from 20-267 mg/l (Appendix VI).

The aerators were installed in Oct89. The effect of the aerators on the SS of the facultative ponds in the period Oct89-Mar90 was evaluated using the routine monitoring data. The levels measured in the effluent from the facultative ponds averaged 152 mg/l and ranged from 80-315 mg/l. This represents a significant increase of 20% over the previous average (127 mg/l) before aeration. However, the results fall in the range of
100-350 mg/l which was determined by Canter and Englande (1970) to be the mean for facultative pond effluents.

Fig 5.15 Weekly suspended solids data from the maturation ponds for the period Mar88-Mar90 (Phase I).

The final effluent of the last maturation pond 2.2, averaged SS of 112 mg/l (range 22-311 mg/l) for the same period. This was higher by 27% than the average before aeration (88 mg/l). Benefield and Randall (1980) report that SS of 150-350 mg/l may be expected in aerobic ponds. The levels found in the final maturation pond during this monitoring phase generally fall within that range, however if compared to the requirements of the Royal Commission Standards (SS = 30 mg/l), the final effluent would not be acceptable for discharge. Although it is difficult to determine if the aerators 'alone' increased the SS of the facultative ponds and through to the final effluent, it is quite reasonable to assume that there appears to have been some influence.

The data in Table 5.7 show that removal of solids at each stage of the WSP system were poor during the monitoring period examined in this chapter.
Table 5.7 Suspended solids percentage removals in WSP system for the periods prior to aerators' installation and after.

<table>
<thead>
<tr>
<th>PERCENTAGE SUSPENDED SOLIDS REMOVALS</th>
<th>Mar88-Aug88</th>
<th>Sep88-Sep89</th>
<th>Oct89-Mar90</th>
</tr>
</thead>
<tbody>
<tr>
<td>PD1.1 Facultative Pond</td>
<td>-11</td>
<td>26</td>
<td>19</td>
</tr>
<tr>
<td>PD1.2 Facultative Pond</td>
<td>7</td>
<td>28</td>
<td>25</td>
</tr>
<tr>
<td>PD2.1 Maturation Pond</td>
<td>57</td>
<td>8</td>
<td>10</td>
</tr>
<tr>
<td>PD2.2 Maturation Pond</td>
<td>16</td>
<td>24</td>
<td>18</td>
</tr>
<tr>
<td>Overall removal</td>
<td>64</td>
<td>49</td>
<td>43</td>
</tr>
</tbody>
</table>

During the start-up period, there was an average increase in the suspended solids in facultative pond 1.1 which resulted in a negative overall removal. Facultative pond 1.2 showed slightly better removals for the same period, 7%. The first maturation pond 2.1 was the most efficient at removing suspended solids with an average removal of 57%, the final maturation pond 2.2 removed solids an average of 16%. The WSP system removed an overall average of 64% during this period. During the Sep88-Sep89 period (before aerators were installed and salinity was increasing), average SS removal in the 2 facultative ponds improved to 26 and 28% respectively, while removal in the first maturation pond declined to 8%. Average removal in the final maturation pond improved during this period to 24% but the average overall removal for the WSP system was down to 49%.

Stabilisation ponds contrast with conventional sewage treatment in that SS are efficiently removed at the primary stage of settlement. When there is aeration at the second stage of biological treatment, more biomass is produced. This is effectively settled because it is mainly bacterial floc. In comparison maturation ponds normally produce algal biomass which is carried over with final effluent therefore aerated WSP systems cannot be expected to achieve significant levels of SS removal.

5.6.2 Dissolved Oxygen (DO) Measurements

Measurement of DO in the ponds commenced in Nov88 after odour problems associated with increased salinity of the sewage occurred. Unfortunately, it was not possible to continue DO measurements, due to laboratory workload, except for the first 2 months of operation of aerators in the facultative ponds. However since there are data available in the first critical months, some observations and discussions follow.
If there is zero or negligible dissolved oxygen in the upper layer of the facultative ponds, the development of anaerobic activity is favoured to occur. This, added to the release of odour-offensive gases produced in the lower (anaerobic) layers of the ponds, generally results in offensive odours emanating from the pond surface.

In the facultative pond 1.1 effluent, the average DO for the period Nov88-Sep89 (before installation of aerators), was 2.2 mg/l, varying between 0-20 mg/l (Fig 5.16). In the cooler months (average pond - 26.6°C) of that period (Nov88-Apr89), no DO was detectable in this pond. However in the hotter summer months (average pond temperature - 30°C) DO was detectable on a number of occasions (Fig 5.16).

Facultative pond 1.2, followed a similar pattern of no detectable DO in the cool months before installation of the aerators and measurable levels in the hot summer months when the temperature in the pond averaged 30°C. The average effluent DO for this period
was 1.4 mg/l. This is lower than that in facultative pond 1.1 for the same period. After the installation of the aerators in Oct89, (Table 3.4, pg 58) there was as expected a marked increase in the DO in both facultative ponds. The average DO in facultative pond 1.1 during the first 2 months of the aerators being used (24 hours/day) was 5.4 mg/l and varied between 0-9.2 mg/l. In facultative pond 1.2, the DO concentration averaged 6.4 mg/l, ranging between 3.2-9.4 mg/l, for the same period (Fig 5.16).

From Fig 5.17, the DO pattern was slightly different in the maturation ponds with detectable concentrations throughout the cool and warm seasons. Before the aerators were installed in the facultative ponds, the average DO measured in the effluent from maturation pond 2.1 was 2.4 mg/l, while in pond 2.2 the average was about the same, 2.6 mg/l.

Fig 5.17 Weekly dissolved oxygen in maturation ponds before and after the installation of aerators in facultative ponds.

After the aerators were installed in Oct89, there was an appreciable increase in the DO concentrations in the maturation ponds. The most significant improvement was in the
final pond 2.2 where the average DO increased from 2.6 to 7.0 mg/l, an increase of >130%. It should be noted, however, that faecal coliform % removal improved in maturation pond 2.1 but reduced in pond 2.2 during the same period.

5.7 Other Parameters Monitored
Other parameters such as sulphate, hydrogen sulphide, ammonia, were analysed as part of the monitoring programme. Some of the results obtained are reported and discussed briefly in this section.

5.7.1 Sulphate (SO₄) and Hydrogen Sulphide (H₂S) Levels in WSP
Due to the problems with salinity and the accompanying sulphate levels, the behaviour of the ponds in terms of sulphur transformations were investigated. Consequently, analyses of raw sewage and pond effluents for sulphate and hydrogen sulphide commenced in Feb89.

The plot, Fig 5.18, demonstrates that the SO₄ levels in the raw sewage more than doubled from the time analyses commenced and up to Mar90. This increase was associated with the time when the EC began rising. The average concentration in the raw sewage for the period was 255 mg/l. During this period there was no significance difference between the concentrations in the primary and maturation ponds where the averages were between 196-204 mg/l.

In the Dec89-Mar90 period, the actual concentrations in the facultative ponds and the first maturation pond exceeded the concentrations in the raw sewage. This may be attributable to aeration inhibiting anaerobiosis and reversing sulphate reduction conversions (Chapter 8). Effluent SO₄ concentrations from both facultative ponds increased >10 fold during the period Nov88-Mar90, from 47.0 to 511.0 mg/l.

The results of the monthly average sulphate concentrations for the WSPs and incoming sewage are shown in Fig 5.18:
Fig 5.18 Monthly average sulphate concentrations in sewage and pond effluents, Nov88-Mar90.

Between Feb89-Sep89 average SO₄ was being removed by microbial conversion to sulphide and sulphur principally in the facultative ponds. No significant additional conversion occurred in the maturation ponds but free H₂S persisted throughout both maturation ponds (Fig 5.19).

After the aerators were commissioned (Oct89-Dec89), SO₄ reduction appears to have been inhibited. However in Jan90, reduction of SO₄ was observed in all ponds but was again inhibited in Feb90 and Mar90 when concentrations measured were higher in the ponds than that measured in the incoming sewage.

For the period (Feb89-Mar90), the H₂S in the raw sewage averaged 9.6 mg/l, ranging from 6-13 mg/l (Fig 5.19). The facultative ponds’ effluents averaged between 1.3 mg/l and 1.6 mg/l and ranged from 0-6 mg/l. In the maturation ponds, H₂S was detectable, however at lower levels than the facultative ponds. The average concentrations in pond 2.1 and pond 2.2, were 0.5 mg/l and 0.2 mg/l, respectively.
For the first 2 months after installation and commissioning of the aerators in the facultative ponds, there was some reduction in H₂S in those ponds. However, in Jan90, the H₂S concentrations began to rise in all ponds (further discussion in Chapters 6, 7 and 8), this coincides with a the reduction in the length of time (from 24 hours/day to 16 hours at night) the aerators were on each day (Table 3.4 and 3.5, pg 58).

![Graph showing monthly average hydrogen sulphide concentrations in raw sewage and pond effluents, Feb89-Mar90.](image)

Surprisingly, positive effects of the aerators in terms of H₂S reduction, were not evident in the effluents from the lagoons. The most likely reason for this is that the draw-off levels from these lagoons is 35 cm from the bottom (Chapter 3) and effluent taken from that depth of the pond is not expected to be from the aerobic stratum.

The following Table 5.8, shows that average H₂S concentrations in the effluent from facultative and maturation ponds increased during the period after the aerators were installed.
Table 5.8 Average hydrogen sulphide concentrations in raw sewage and pond effluents, before and after aerators were commissioned in facultative ponds.

<table>
<thead>
<tr>
<th></th>
<th>Raw sewage</th>
<th>PD 1.1</th>
<th>PD 1.2</th>
<th>PD 2.1</th>
<th>PD 2.2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Before aerators</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Feb89-Sep89</td>
<td>9.8</td>
<td>1.5</td>
<td>1.3</td>
<td>0.03</td>
<td>0.0</td>
</tr>
<tr>
<td>After aerators</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Oct89-Mar90</td>
<td>9.3</td>
<td>1.7</td>
<td>1.2</td>
<td>1.03</td>
<td>0.5</td>
</tr>
</tbody>
</table>

Further discussions on the correlations between salinity, SO₄ and H₂S concentrations in the raw sewage are located in Chapter 8.

5.7.2 Ammonia-Nitrogen (NH₃-N) Removals

The ammonia and organic forms of nitrogen are the main sources of nitrogen in WSP systems. Various investigators have reported that nitrification in sewage treatment ponds does not proceed at an appreciable level or consistently (Oswald et al, 1970; US EPA, 1975; Yanez et al, 1980; and WHO, 1987b). This is most likely due in part to inhibition of nitrification due to toxic substances.

Facultative ponds are the most effective of all biological treatment processes in the removal of nitrogen although all the mechanisms involved are unclear according to Horan (1990). This is in contrast to James (1987) who states that the efficiency of WSPs in removing nutrients is poor because much of the nitrogen and phosphorus leaves the pond system in the form of algae. This is a matter of opinion because uptake of nitrogen by algae and bacteria is viewed by others (Santos and Oliveira, 1987) as one process whereby nitrogen is removed in the ponds. Mara et al (1992b) noted that although the removal of nitrogen in WSPs is good, very little data are available on nutrient removal.

There are wide variations in the removal of ammonia reported in ponds, from negligible (Toms et al, 1975; and Silva et al, 1987) to values as high as 95% (Middlebrooks et al, 1982). The higher removals reported were associated with retention times of >30 days. Silva et al (in press) reported 92% removal in a 23 days retention time pond system (raw sewage-32.5 mg/l) to 2.6-4.3 mg/l in maturation ponds. Mara et al (1992b) reported that in German rural communities, maximum ammonia removal of about 75% was experienced in typical WSPs.
Although as previously mentioned, some of the literature reviewed suggested the removal mechanisms of nitrogen compounds in sewage ponds are unclear, Mara et al (1992b) explained that it is relatively simple. Mara et al (1992b) do not accept that the nitrification/denitrification processes occur to any great significance because the environment necessary (aerobic surface area for attached growth) for the establishment of a strong community of nitrifying bacteria is not typically available. In fact these authors suggest that major mechanisms for the removal of nitrogen from WSPs is either through algal uptake or volatilisation at the pond surface. Muttamara and Puerpaiboon (in press) found in lab-scale experiments that biological uptake was the major nitrogen removal mechanism. Ammonia volatilisation was also found to make a considerable impact on the removal of nitrogen.

Santos and Oliveira, (1987), in contrast, submit that there are four major processes generally accepted as influencing the nitrogen cycle in WSP systems:

1. Biological hydrolysis of organic nitrogen, releasing NH$_3$-N.
2. Nitrogen assimilation by bacteria and algae, building up cell biomass
3. Nitrification of NH$_3$, producing NO$_2$ and NO$_3$.
4. Denitrification of NO$_3$, releasing N$_2$.

Ponds, especially facultative ponds simultaneously exhibit aerobic and anaerobic behaviour. This would suggest that nitrification followed by denitrification is the main removal mechanism of nitrogen. However, Horan (1990) noted that isolation of nitrifying bacteria in ponds revealed low numbers present, thereby supporting the hypothesis of Mara et al (1992b). Organic nitrogen associated with biomass will be bound in the bottom sludge accounting for some removal. Additionally some of the ammonium produced in the sludge layer (by anaerobic degradation of organic material) may be converted to ammonia gas and volatilised to the atmosphere. This will occur if there is an active algal band (algae will also take up NH$_4$) in the pond resulting in high pH’s according to the following reaction:

\[
\text{NH}_4^+ + \text{OH}^- \overset{\text{pH} > 9.0}{\iff} \overset{\text{pH} < 9.0}{\text{NH}_3 + \text{H}_2\text{O}}
\]

Eq. 5.1

In nitrification, the removal of ammonia and ammonium ions in ponds is normally a two stage process. The two stages are distinct, each involving different species of
chemo-autotrophic nitrifying bacteria. The nitrifying bacteria utilise ammonia or nitrite as an energy source. This first stage of ammonium oxidation to nitrite is termed nitrosification and is represented by the following reaction:

\[ \text{NH}_4^+ + 1.5\text{O}_2 \rightarrow \text{NO}_2^- + 2\text{H}^+ + \text{H}_2\text{O} \quad \text{Eq. 5.2} \]

*Nitrosomonas* bacteria are generally considered to be the catalyst for this reaction, the first stage of nitrification. During the oxidation of ammonia to nitrite, hydrogen ions are released resulting in a lowering of the pH of the wastewater. In WSP systems which are characterised by long retention times this will influence the development of algal activity.

In the second stage of nitrification, nitrite is oxidised to nitrate. This is represented by the following reaction:

\[ \text{NO}_2^- + 0.5\text{O}_2 \rightarrow \text{NO}_3^- \quad \text{Eq. 5.3} \]

The bacteria considered to be responsible for the second stage of nitrification are the *Nitrobacter* sp. Anaerobic conditions will inhibit nitrifying bacteria as they are strict aerobes (Gray, 1989). Therefore as oxygen in the Cayman Islands ponds is very low due to the overproduction of hydrogen sulphide, this process is clearly limited or non-existent. This is observed in that there is little consequential reduction of ammonia-nitrogen in the maturation ponds.

Because both aerobic and anaerobic metabolism take place in WSP systems as opposed to most conventional biological processes, under appropriate conditions the nitrogen cycle may be completed (Fig 5.20). Rapid conversion to nitrogen by denitrifying bacteria in the anoxic and anaerobic layers found in sewage treatment lagoons most likely accounts for the fact that high concentrations of nitrates are not commonly found in sewage treatment ponds (WHO, 1987b).
Nitrogen ($N_2$)

**Denitrification (Anoxic)**

Nitrates ($NO_3^-$)

**Nitrogen Fixation (Anaerobic)**

Ammonia ($NH_3$)

**Ammonification**

proteins etc microbes, plants & animals

**Assimilation (Aerobic)**

Nitrites ($NO_2^-$)

**Assimilation**

**Nitrification**

Fig 5.20 Simplified nitrogen cycle showing interconversions which can occur in aquatic environments (adapted from Horan, 1990).

However in Mexico, Rivera et al (1986) reported significant nitrate levels of 0.10-1.39 mg/l in a small pond system (serving 1200 population). They attributed this to the nitrifying processes taking place in the surface strata of the ponds. In contrast, Santos and Oliveira (1987), in a study carried out in ponds in Portugal, reported that in a facultative pond with 17.3 days retention time that nitrification was more significant in ammonia removal (bacteria and algae assimilation and nitrifying bacteria) than denitrification. In addition, they reported that their results could not identify the denitrification process occurring.

Denitrification occurs in the anoxic sediment/sludge layer and occurs more slowly under acid conditions compared with neutral or alkaline conditions. Although small amounts of nitrates are used by WSP algae for normal nutritional requirements, most of the denitrification occurs at the sludge-water interface (Toms et al, 1975).

The average concentration of NH$_3$-N in the Cayman Islands raw sewage samples (24 hour composite since Sep89) was 49 mg/l which falls in the range of typical concentrations found in sewage (Ruffier et al, 1981; Gray, 1989; and Horan, 1990) although higher than the average 29.3 mg/l reported in raw sewage entering the San Juan ponds in Peru (Yanez et al, 1980). The average concentrations in the raw sewage
remained stable, between 50-60 mg/l, until after Oct89 when the levels dropped significantly (Fig 5.21). This dramatic drop in the raw sewage ammonia concentrations is unexplainable. However, normal concentrations returned in the sewage after 2 months.

The average results from the individual pond effluents indicate that ammonia-nitrogen transformations take place mainly in the facultative ponds as ammonia concentrations remain fairly consistent in the effluents from the 4 ponds (Fig 5.21).

![NH3-N](image)

**Fig 5.21** Monthly mean ammonia-nitrogen concentrations in raw sewage and in pond effluents at the sewage treatment works Mar88-Mar90.

The removal of ammonia in the WSP for the monitoring period May89-Mar90 was 72%. The theoretical retention time of the system during that period was 32 days. These results, when compared with other reports, indicate that the Cayman Islands ponds were not very efficient in removing ammonia, even with the long retention time.
It is reasonable then, to expect poor nitrification in facultative ponds if oxygen is limiting and \( \text{H}_2\text{S} \) is present. Nitrification is a sensitive microbiological process, susceptible to inhibition, which is clearly going on in the maturation ponds, as well. Fig 5.21 visually demonstrates the impact of aerators on ammonia conversion comparing ponds' effluent results prior to Oct89 (before installation of aerators) and during the period Oct89-Mar90 (after installation of aerators).

5.8 General Meteorological Conditions

The important influence of environmental conditions on the performance of WSP systems was clearly indicated in the literature review (Chapter 2). Consequently meteorological data collection is included as a part of the Cayman Islands sewage treatment ponds monitoring programme. This thesis will not address in depth all aspects of climatic data for which averages may be presented.

The meteorological conditions during the monitoring period Mar88-Mar90, did not deviate from the norm expected for the Cayman Islands. That is with the exception of hurricane Gilbert in Sept88, which brought unusually high windspeeds and flooding with seawater of the main sewers and interceptors.

The average daily rainfall and the total rainfall data for the year, 1988 was obtained from the Civil Aviation Meteorological Station at the Owen Roberts International Airport, Grand Cayman. The station is located approximately 2 km from the sewage treatment works. In 1989, collection of rainfall data on-site commenced, while in 1990, daily windspeed recording began.

Basic meteorological data for the years 1988 and 1989 are summarised in Table 5.9:
Table 5.9 Annual averages and seasonal variations in climatic data for 1988 and 1989.

<table>
<thead>
<tr>
<th>METEOROLOGICAL DATA</th>
<th>1988</th>
<th>1989</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rainfall in mm</td>
<td>Met station</td>
<td>STW</td>
</tr>
<tr>
<td>total rainfall</td>
<td>1354.1</td>
<td>1305.2</td>
</tr>
<tr>
<td>annual daily average</td>
<td>3.71</td>
<td>3.58</td>
</tr>
<tr>
<td>daily average, dry season (dec-apr)</td>
<td>1.15</td>
<td>1.56</td>
</tr>
<tr>
<td>daily average, wet season (may-nov)</td>
<td>5.52</td>
<td>5.00</td>
</tr>
<tr>
<td>Wind direction in degrees ° from north</td>
<td></td>
<td></td>
</tr>
<tr>
<td>annual average</td>
<td>99.2</td>
<td>97.5</td>
</tr>
<tr>
<td>average, dry season (dec-apr)</td>
<td>90.0</td>
<td>96.0</td>
</tr>
<tr>
<td>average, wet season (may-nov)</td>
<td>105.7</td>
<td>98.6</td>
</tr>
<tr>
<td>Windspeed m/sec</td>
<td></td>
<td></td>
</tr>
<tr>
<td>average</td>
<td>5.14</td>
<td>3.60</td>
</tr>
<tr>
<td>average, dry season (dec-apr)</td>
<td>3.50</td>
<td>3.90</td>
</tr>
<tr>
<td>average, wet season (may-nov)</td>
<td>4.06</td>
<td>3.40</td>
</tr>
<tr>
<td>Ambient air temperature °C</td>
<td></td>
<td></td>
</tr>
<tr>
<td>average</td>
<td>27.8</td>
<td>27.4</td>
</tr>
<tr>
<td>dry season (dec-apr)</td>
<td>26.4</td>
<td>26.1</td>
</tr>
<tr>
<td>wet season (may-nov)</td>
<td>28.7</td>
<td>28.4</td>
</tr>
</tbody>
</table>

5.9 Summary

The following observations are made in summation of the results of this monitoring phase:

1. A protracted period of increasing flow and salinity (17 months) was recorded and attributed to increasing problems of groundwater intrusion due to 'deterioration/breakdown' of sewerage pipes.

2. Flow to the sewage treatment works at the end of this period (Mar88-Mar90) was already approaching that of the design flow (expected in 1996). The nominal retention time for the WSP system was 40 days.

3. By the end of Mar90, salinity increased to >10000 μS/cm from an average of 2205 μS/cm (average before hurricane in Sep88). This indicates that a major % of flow was attributable to saline groundwater intrusion.

4. The BODuf of the incoming sewage was reduced over the last 14 months of this study phase due to dilution. The reduced BODuf removals were attributed principally to the dilution effect. Overall BODuf % removal is similar in the two later phases, in spite of lower BODuf loading during the last period: but in the last
period (Sep90-Mar90) the absolute values of BODuf leaving the maturation ponds are lower than in Oct88-Aug89 and this may be attributable to artificial aeration in the facultative ponds.

5. The removal of faecal coliforms in the facultative ponds is impressive in the face of the increasing flow; however this is less so in the maturation ponds. Overall performance did not achieve the guidelines of irrigation reuse in spite of the baffles intervention. The influence of H$_2$S is probably unimportant in the facultative ponds, but damaging the performance in the maturation ponds since they (the ponds) were anoxic for protracted periods.

6. Although the facultative ponds are designed to treat relatively weak sewage (low BOD loadings) it could not have been foreseen that this would have arisen from groundwater intrusion with the attendant high SO$_4$. This was conducive to the production of excessive sulphide (H$_2$S) in the facultative ponds and a carry-over to the maturation ponds. The effect of this has been to reduce their performance for faecal coliform removal through the inhibition of aerobic processes (photosynthesis and predation).

7. The priorities for monitoring in phase II, were expansion of the programme to include sludge depth measurements every 6 months, inclusion of volatile and fixed suspended solids measurements, and continued improvements in analytical methods.
CHAPTER 6

6.0 ROUTINE MONITORING PHASE II APRIL 1990 TO MARCH 1993

6.1 Monitoring of WSP System April 1990 - March 1993
Monitoring of the WSP system in Cayman continued and was further expanded during the period Apr90-Mar93 to include sludge depth every 6 months, volatile and fixed suspended solids, diurnal DO and temperature, and collection of windspeed data. Regular monitoring of top water level in each pond at the time of sample collection was included in the programme. In support of another research project, monitoring of weekly maximum and minimum pond temperatures was carried out for a 1 year period (Jan91-92).

Several short-term investigative studies were also carried out during this monitoring phase as part of the research for this thesis:

1. Biological monitoring of the phyto and zooplankton in the ponds commenced and included measurement of chlorophyll a concentrations.

2. A preliminary survey of ova/parasites in raw sewage was carried out using a simple technique.

3. A faecal coliform bacteria die-off study was performed in a batch study to establish/determine appropriate \( k_b \) values for the sewage treatment lagoons.

The above experiments and their results are described in Chapter 9.

The performance of the sewage ponds from Apr90-Mar93 was again largely influenced by increased flows and salinity. Remediation and replacement of electrical control panels, relining of several broken sewer mains, and relining of pump stations and manholes commenced or were completed during this period.

6.2 Salinity and Increased Flows
During the 35 months of routine monitoring in phase II, sewage inflow and salinity continued to increase as shown in Fig 6.1. By the end of the phase both the EC and
inflow had almost doubled from the means reported in Chapter 5 for the last 6 months of phase I.

Salinity and inflows in phase II may be assessed in 2 distinct periods, from Apr90-Dec90 and Jan91-Mar93. These 2 periods showed wide variations in daily average inflow and EC as shown in Fig 6.1. The period Apr90-Dec90 was characterised by a mean daily inflow <3000 m³/day whereas in Jan91-Mar93 the mean daily inflow to the works was just over 4300 m³/day, that is, 50% (overload) above the maximum design flow. For the same periods, average EC was 12768 μS/cm and 19435 μS/cm, respectively.

The hypothesis tested in Chapter 5 where it was proposed that if infiltrating saline groundwater caused increased flows then the correlation of flow and salinity should produce a high $R^2$. This assumption was tested again in Fig 6.2 and was confirmed with a reasonably significant relationship of $R^2 = 0.7$. 

Fig 6.1 Average daily incoming sewage flow and electrical conductivity for the period Apr90-Mar93 (Phase II).
At the peak of high salinity in the sewage it is estimated that up to 45% of flow was due to saline groundwater infiltration. For the period Apr90-Mar93, chloride levels in composite samples of raw sewage averaged 6302 mg/l with results ranging from 3060 to 9847 mg/l. As a result the final effluent remained unsuitable for irrigation use during the entire monitoring phase.

The consequences of saline sewage were increased pumping costs, increased maintenance and operational costs, reduced treatment efficiency, and corrosive stress on structures (further discussed in Chapter 8). Clearly it was in the best interest of the Authority to endeavour to repair the failing structures in the sewerage system. Major repairs on 3.7 km of 150 mm clay pipes commenced in Jan93 and were completed by the end of Mar93. The dramatic effect of the repairs is seen in Fig 6.1 where the inflow decreased from a mean of 5000 m³/day in Feb93 to an average of just over 3000 m³/day in Mar93.
Mean retention time throughout the treatment system was also affected by the increased flows in the Apr90-Mar93 period. The average nominal retention of both facultative ponds was reduced to only 8 days, that of each maturation pond 2.1 and 2.2 to 3 days. Consequently average retention time throughout the entire WSP system was reduced by approximately 65% of the average reported in the previous chapter, that is from 40 days to 14 days. The nominal retention times reported here do not take into account the volume unavailable due to sludge accumulation and only take into account the top water level.

6.2.1 Organic Loading, BOD and COD Removals

Unfortunately, no measurement of BOD data was possible from Sep92-Jun93 because the BOD incubator was not functional and funds were not available for a replacement until Jun93. The raw sewage BOD uf during the period Apr90-Aug92 averaged 80 mg/l and ranged from 38.6 mg/l to 146.0 mg/l (Fig 6.3 and Table 6.1).

![Fig 6.3 Monthly mean BOD uf of raw sewage inflow, facultative ponds and percentage removals in the facultative ponds during the period Apr90-Mar93 (Phase II).](image-url)
From Fig 6.3 and Table 6.5, BODuf removal in the facultative ponds averaged 58% down significantly from the removal efficiency reported (75%) in the same ponds for the last 6 months of phase I (Chapter 5).

Table 6.1 BODuf concentrations and loadings on facultative and maturation ponds Apr90-Mar93.

<table>
<thead>
<tr>
<th></th>
<th>BODuf mg/l</th>
<th>BODuf loading kg/ha day</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
<td>Max</td>
</tr>
<tr>
<td>In Sew</td>
<td>80.1</td>
<td>146.0</td>
</tr>
<tr>
<td>PD 1.1</td>
<td>32.7</td>
<td>84.2</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>34.8</td>
<td>76.5</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>24.1</td>
<td>47.3</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>21.4</td>
<td>42.4</td>
</tr>
</tbody>
</table>

The majority of the work is being done by the facultative ponds (59% reduction), whereas BODuf removal in the first maturation pond 2.1 averaged 28.5% meanwhile the final maturation pond 2.2 averaged only 11% (Table 6.5). Nonetheless, the overall system removal for the period met the BOD standard of 30 mg/l.

Fig 6.4 Monthly mean BODuf of effluent from maturation ponds and overall system percentage removals during the period Apr90-Mar93 (Phase II).
The overall BODuf removal efficiency of the WSP system for the entire period Apr90-Aug92 was 73% (Fig 6.4), a dramatic decrease of 14% from the previous period described in Chapter 5.

The soluble BODf of the raw sewage averaged 41.3 mg/l (Table 6.2) and contributed to 52% of the total average BODuf. In the facultative ponds the contribution of soluble BOD to the total BOD was 60-64%. In the maturation ponds soluble BOD contribution was 42-46%.

Table 6.2 BODf concentrations and loadings on facultative and maturation ponds Apr90-Mar93.

<table>
<thead>
<tr>
<th></th>
<th>BODf mg/l</th>
<th></th>
<th>BODf loading kg/ha day</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
<td>Max</td>
<td>Min</td>
<td>Mean</td>
</tr>
<tr>
<td>In Sew</td>
<td>41.3</td>
<td>92.8</td>
<td>21.8</td>
<td>n/a</td>
</tr>
<tr>
<td>PD 1.1</td>
<td>19.7</td>
<td>51.9</td>
<td>8.0</td>
<td>88.1</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>22.1</td>
<td>43.0</td>
<td>9.4</td>
<td>88.1</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>11.2</td>
<td>29.5</td>
<td>3.7</td>
<td>60.8</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>9.0</td>
<td>20.0</td>
<td>4.4</td>
<td>31.9</td>
</tr>
</tbody>
</table>

The data show that organic loading in terms of BOD steadily decreased as the flow increased. Although the WSP system was hydraulically overloaded due to infiltration, BOD loading was well below that used in the design of the ponds.

In contrast with the period discussed in the previous chapter, removal efficiencies correlations between BODuf removals and EC were not significant indicating that removal is more dependent on flow than salinity.

6.2.2 COD Loading and Removal

The incoming sewage CODuf values varied from 259 to 733 mg/l and those of the final effluent from 184 to 498 mg/l (Table 6.3). CODuf removal in the facultative ponds averaged 25%, a dramatic reduction in performance compared to the removal efficiency (47.5%) reported in phase 1 (Chapter 5). The maturation ponds 2.1 and 2.2 averaged 8% and 2% respectively (Table 6.5). Maturation pond 2.1 removal efficiency dropped
significantly compared to phase 1 where average removal was 54%. There was little change in the average removal in pond 2.2, the final maturation pond.

Table 6.3 CODuf concentrations and loadings on facultative and maturation ponds Apr90-Mar93.

<table>
<thead>
<tr>
<th></th>
<th>Mean</th>
<th>Max</th>
<th>Min</th>
<th>CODuf loading kg/ha day</th>
</tr>
</thead>
<tbody>
<tr>
<td>In Sew</td>
<td>514.2</td>
<td>732.6</td>
<td>259.2</td>
<td>n/a</td>
</tr>
<tr>
<td>PD1.1</td>
<td>351.2</td>
<td>594.0</td>
<td>175.8</td>
<td>1107.7 1827.4 758.0</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>350.0</td>
<td>531.0</td>
<td>217.3</td>
<td>1104.6 1817.6 758.9</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>323.1</td>
<td>477.7</td>
<td>188.6</td>
<td>1050.0 1929.0 635.7</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>317.6</td>
<td>497.7</td>
<td>183.7</td>
<td>996.6 1668.4 456.2</td>
</tr>
</tbody>
</table>

During this monitoring period the CODuf surface loading on the facultative ponds averaged about 1100 kg/ha day. The maturation ponds 2.1 and 2.2 received on average 1050 kg/ha day and 997 kg/ha day, respectively (Table 6.3).

The CODf of the influent showed wider variations and higher concentrations during this monitoring phase than in the preceding one, ranging from 219 to 600 mg/l and averaged about 360 mg/l representing 70% of the total COD (Table 6.4).

Table 6.4 CODf concentrations and loadings on facultative and maturation ponds Apr90-Mar93.

<table>
<thead>
<tr>
<th></th>
<th>Mean</th>
<th>Max</th>
<th>Min</th>
<th>CODf loading kg/ha day</th>
</tr>
</thead>
<tbody>
<tr>
<td>In Sew</td>
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<td>601.4</td>
<td>219.2</td>
<td>n/a</td>
</tr>
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<td>PD1.1</td>
<td>243.1</td>
<td>373.3</td>
<td>136.5</td>
<td>787.6 1530.8 398.7</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>250.2</td>
<td>402.5</td>
<td>127.5</td>
<td>786.3 1522.6 366.8</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>195.0</td>
<td>373.8</td>
<td>91.5</td>
<td>739.0 1367.2 422.0</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>187.8</td>
<td>367.9</td>
<td>101.0</td>
<td>620.7 1334.2 203.0</td>
</tr>
</tbody>
</table>
The CODf of the final effluent varied between 101 and 368 mg/l with an average of 188 mg/l. CODf removal efficiency in the facultative ponds was about the same as CODuf removal (Table 6.5). Removal in maturation ponds 2.1 and 2.2 was 21% and 4%, respectively. During this period CODf surface loading on the facultative ponds averaged 787 kg/ha day and was 739 kg/ha day and 621 kg/ha day on the maturation ponds 2.1 and 2.2, respectively.

Table 6.5 BOD and COD removal efficiency in facultative and maturation ponds Apr90-Mar93.

<table>
<thead>
<tr>
<th></th>
<th>Percentage Removal</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>BODuf</td>
</tr>
<tr>
<td>PD1.1</td>
<td>59.2</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>56.6</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>28.5</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>11.1</td>
</tr>
<tr>
<td>Overall % removal</td>
<td>73.3</td>
</tr>
</tbody>
</table>

Both CODuf and CODf removal efficiencies in facultative ponds and especially in the maturation pond 2.1 were low during this period. Increasing COD in the raw sewage and across the ponds is attributed to the inorganic oxygen demand of basically anaerobic pond effluents and the elevated sulphide levels found through the system. Overall CODuf and CODf removal was 32% and 42% respectively for phase II monitoring period.

6.2.3 Increasing Salinity and Faecal Coliform Removal Efficiencies
As observed in Chapter 5, elevated salinity in the raw sewage appeared to negatively influence the faecal coliform removal efficiencies in the WSP system. Examination of the faecal coliform data (Fig 6.5) during the 2 periods of distinct salinities described earlier in this chapter further illustrate the effect of these operating conditions.
In Apr90-Dec90 when the average daily flow was approximately 3000 m³/day and EC averaged about 12800 μS/cm, faecal coliform density in the raw sewage was $1.03 \times 10^8$ cfu/100 ml (Table 6.6). There was an average of 2 log removal in the facultative ponds, meantime in maturation ponds 2.1 and 2.2, 81 and 76% were removed, respectively. Overall faecal coliform removal during this period averaged 4 logs and was similar to that reported in the preceding chapter in the last 6 months of phase I (Fig 5.13, pg 92). At no time was the required effluent standard of <1000 faecal coliforms cfu/100 ml met.

Table 6.6 Comparison of faecal coliform removal efficiencies during periods of differing salinity and average daily flow.

<table>
<thead>
<tr>
<th></th>
<th>Apr90-Dec90</th>
<th>Jan91-Mar93</th>
<th>Apr90-Mar93</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>avg daily flow 2980 m³/day</td>
<td>avg daily flow 4380 m³/day</td>
<td>avg daily flow 4030 m³/day</td>
</tr>
<tr>
<td></td>
<td>avg EC 12768 μS/cm</td>
<td>avg EC 19435 μS/cm</td>
<td>avg EC 17768 μS/cm</td>
</tr>
<tr>
<td></td>
<td>Mean</td>
<td>% removal</td>
<td>Mean</td>
</tr>
<tr>
<td>In Sew</td>
<td>$1.03 \times 10^8$</td>
<td>n/a</td>
<td>$2.17 \times 10^6$</td>
</tr>
<tr>
<td>PD 1.1</td>
<td>$1.89 \times 10^7$</td>
<td>99.82</td>
<td>$1.26 \times 10^5$</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>$1.36 \times 10^7$</td>
<td>99.87</td>
<td>$1.08 \times 10^4$</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>$2.98 \times 10^6$</td>
<td>81.66</td>
<td>$2.52 \times 10^3$</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>$7.12 \times 10^5$</td>
<td>76.13</td>
<td>$9.57 \times 10^2$</td>
</tr>
<tr>
<td>overall</td>
<td>n/a</td>
<td>99.993</td>
<td>n/a</td>
</tr>
</tbody>
</table>
In the period Jan91-Mar93, the diluting effect of groundwater infiltration became more evident as the mean faecal coliform density measured in the raw sewage decreased by more than 1 log (Table 6.6). Faecal coliform reduction in the facultative ponds was reduced by 1 log during this period although for the entire 35 months of phase II, overall % removal in the facultative ponds averaged 99.5%. In maturation ponds 2.1 and 2.2, removals were poor at 78 and 62% respectively. While, for the duration of phase II, overall faecal coliform reduction in the WSP system was on average 3 logs and the effluent standard was met in only one month (Dec91).

pH levels of the raw wastewater and the WSP for the monitoring period under discussion are summarised in Table 6.7. Generally pH levels increased from facultative ponds through to maturation ponds although the high levels reported for the period prior to saline intrusion (Table 5.2, pg 94) were not observed in the Apr90-Mar93 monitoring programme.

<table>
<thead>
<tr>
<th></th>
<th>pH units</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
</tr>
<tr>
<td>IN SEW</td>
<td>7.54</td>
</tr>
<tr>
<td>PD 1.1</td>
<td>7.87</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>7.83</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>7.96</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>7.97</td>
</tr>
</tbody>
</table>

As in the foregoing monitoring phase I, algal photosynthesis appears to have been impeded by H$_2$S and thus the high pH levels normally linked to algal activity were not detected in pond effluents. As expected no significant correlation was observed between pH levels and faecal coliform reduction efficiency of the WSP system.

### 6.2.4 Sulphate and Hydrogen Sulphide Concentrations in Ponds

As the sewage flow continued to be affected by infiltration, analysis of SO$_4$ and H$_2$S concentrations through the sewage treatment works continued in phase II. Frederick
(1991a) reported on the continual rise in SO$_4$ and the accompanying high levels of H$_2$S produced in the facultative ponds of the Cayman Islands WSP.

Sulphate concentrations in the sewage inflow to the treatment works increased in the period Apr90-Dec90 to an average of 624 mg/l with approximately 22-35% being removed/reduced in the facultative ponds. Overall conversion during this period averaged 28%. By the end of phase II, SO$_4$ concentration in the raw sewage had more than doubled (Fig 6.6).

Fig 6.6 Monthly average sulphate concentrations in sewage and pond effluents, Apr90-Mar93 (Phase II).

After Dec90 sulphate transformations in both facultative ponds resulted in an average decrease/removal of 9% (Table 6.8). In the maturation ponds much less conversion occurred resulting in only 2% removal. Overall reduction/removal was only 12% reflecting the poor efficiency of the system in handling elevated SO$_4$ concentrations. Removal by way of microbial conversion to hydrogen sulphide and sulphur is considered to be the main mechanism occurring in the facultative ponds. However, the lower % conversion may be attributable to the dilution of organic nutrients from sewage which could now be the limiting factor whereas SO$_4$ is in excess.
Table 6.8 Sulphate concentrations and removal in facultative and maturation ponds Apr90-Mar93.

<table>
<thead>
<tr>
<th></th>
<th>Mean</th>
<th>Max</th>
<th>Min</th>
<th>% removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>IN SEW</td>
<td>1052</td>
<td>1775</td>
<td>453</td>
<td>n/a</td>
</tr>
<tr>
<td>PD 1.1</td>
<td>954</td>
<td>1800</td>
<td>288</td>
<td>9</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>953</td>
<td>1900</td>
<td>318</td>
<td>9</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>939</td>
<td>1560</td>
<td>276</td>
<td>2</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>921</td>
<td>1460</td>
<td>280</td>
<td>2</td>
</tr>
<tr>
<td>overall removal</td>
<td>12</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

For the period Apr90-Mar93, the mean H₂S in the inflow was 5.2 mg/l and ranged from 2.0-10.0 mg/l (Fig 6.7 and Table 6.9). The facultative ponds' effluents exhibited very high H₂S levels, averaging between 14-15.0 mg/l and wide variations from 0-40.0 mg/l.

Fig 6.7 Monthly mean hydrogen sulphide concentrations in sewage and pond effluents, Apr90-Mar93 (Phase II).

This presented a dramatic difference from the levels observed and reported in Chapter 5 for monitoring phase I. In fact, H₂S production in the facultative ponds was such that
on average $\text{H}_2\text{S}$ increased by almost 170%. Of course, these high levels of $\text{H}_2\text{S}$ gave rise to numerous complaints of odour. However, because of high electricity costs, the aerators were only used on 2 occasions (Nov91-Mar92 and Jan93-Mar93) during this period (Fig 6.7). It was only during those 2 short periods that $\text{H}_2\text{S}$ concentrations dropped significantly.

Table 6.9 Hydrogen sulphide concentrations and conversion in facultative and maturation ponds Apr90-Mar93.

<table>
<thead>
<tr>
<th></th>
<th>Mean</th>
<th>Max</th>
<th>Min</th>
<th>% conversion</th>
</tr>
</thead>
<tbody>
<tr>
<td>IN SEW</td>
<td>5.2</td>
<td>10.0</td>
<td>2.0</td>
<td>n/a</td>
</tr>
<tr>
<td>PD 1.1</td>
<td>14.0</td>
<td>40.0</td>
<td>0</td>
<td>-169</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>15.0</td>
<td>35.0</td>
<td>0</td>
<td>-188</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>4.1</td>
<td>22.5</td>
<td>0</td>
<td>71</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>2.5</td>
<td>22.5</td>
<td>0</td>
<td>40</td>
</tr>
<tr>
<td>overall % conversion</td>
<td></td>
<td></td>
<td></td>
<td>52</td>
</tr>
</tbody>
</table>

In the maturation ponds 2.1 and 2.2, lower levels of $\text{H}_2\text{S}$ persisted with mean concentrations of 4.1 mg/l and 2.5 mg/l, respectively. Despite this, the net conversion in the WSP was 52%.

6.2.5 Saline Sewage and Corrosion of Concrete Sewer Structures

The manholes, lift stations and concrete structures for the system suffered deterioration due to the high sulphate content of the sewage. Corrosion of concrete structures is due to the reduction of sulphate to sulphide by anaerobic bacteria which is then used by *thiobacilli* that utilise the hydrogen sulphide. The resulting reaction is the production of sulphuric acid. The sulphuric acid then causes the concrete to corrode as further described in Chapter 8.

The WAC repaired 9 lift/pumping stations and 29 concrete manhole structures that were degraded to unacceptable levels after <6 years of service. This was done using a Permaform system that utilises a corrosion resistant T-lock lining (Shook, 1991).

Another effect of the presence of hydrogen sulphide gas on the sewer system was the failure of the control panels for the 12 lift stations. In 1989, coinciding with the
increasing salinity in the sewage, the control panels began to fail frequently. This was attributed to the corrosive effect of hydrogen sulphide on delicate mechanisms in the control panel. Symptoms observed were erratic tripping of overloaded protective relays, burned fuse holders and brittle copper conductors (Jager, 1991). As a result of this problem, special control panels to withstand the corrosive environment under which they must operate were designed and commissioned by a private company, Polytron, under contract by the WAC. These repairs were completed in 1992 at considerable cost, however WAC maintenance expenses (operator time and energy costs) were substantially reduced.

6.3 Ammonia-Nitrogen Removals
During the period Apr90-Dec90 (when the EC averaged 12768 μS/cm), the mean NH₃-N in the raw sewage was 24.5 mg/l. Overall removal from the system was 47%. Most of the NH₃-N, 31-33%, was removed/converted in the facultative ponds. NH₃-N removal in facultative ponds is likely to be largely due to heterotrophic bacterial uptake. The maturation ponds are not wholly aerobic, so effectively no nitrification takes place as nitrifying bacteria are strictly aerobic. The percentage removal in maturation pond 2.1 was 19% and removal in the final maturation pond 2.2 was only 4% resulting in the final effluent having a mean of 13.0 mg/l NH₃-N. However as the incoming flow increased, the NH₃-N in the raw sewage decreased by 45% reflecting the dilution effect of the infiltrating groundwater (Fig 6.8).

During the period Jan91-Mar93 removal efficiency in the facultative ponds was reduced by almost half to 16-20%, while the maturation ponds showed improved removal efficiencies increasing to 20% and 9% respectively. These results indicate that NH₃-N removal or conversion in the facultative ponds was hindered due to the prevailing anoxic conditions in the maturation ponds.
Fig 6.8 Monthly average ammonia-nitrogen concentrations in raw sewage and in effluent of ponds, Apr90-Mar93 (Phase II).

The mean NH$_3$-N level in the raw sewage for this 35 month period (Apr90-Mar93) was 16.1 mg/l (Fig 6.8 and Table 6.10). The effect of dilution is reflected in this mean result as it is > 3 times less the average reported in phase I (Fig 5.20).

Table 6.10 Ammonia-nitrogen concentrations and removal in facultative and maturation ponds Apr90-Mar93.
Overall NH$_3$-N removal was seriously impaired during this second period (Apr90-Mar93), it dropped to 43% compared to 72% removal in phase I (Fig 5.20, pg 113).

6.4 Aerators in Facultative Ponds

During this period the aerators were used from the end of Nov91 until mid-Mar92. Only 4 were used in each facultative pond and they were turned on at night averaging about 15 hours per night. All functioning aerators (7 in pond 1.1 and 6 in pond 1.2) were put to use for the last time at the end of Jan93 until mid-Mar93. As may be seen from Figs 6.9 and 6.10, the aerators did not dramatically affect DO levels in effluent from the ponds.

It should be noted that effluent samples taken for the regular routine monitoring programme were collected at the weir chambers as described in Chapter 4. During this period the water level in the facultative ponds varied from 1.61 m to 2.10 m, the maturation ponds ranged from 1.34 m to 1.97 m. It follows that an effluent sample taken from the weir chamber is representative of pond liquid at the depth of the outlets (Chapter 3). For that reason, data reported in Figs 6.9-6.10 represent samples from an average depth (from top water level) of 1.4 m in the facultative ponds, 1.2 m in the maturation pond 2.1, and 1.0 m in the final maturation pond, respectively. The DO data reported in the effluents indicate the extent or depth of aerobic activity in these samples at 0900 hours.

Fig 6.9 illustrates that in the facultative ponds, during the hot summer months (May-Nov, with no mechanical aeration), DO levels increased. The maturation ponds followed a similar pattern as the facultative ponds (Fig 6.10). Figs 6.9 and 6.10 show that when used the aerators had little impact on DO levels in the effluent of the facultative ponds.
Fig 6.9 Monthly mean DO concentrations in effluent from facultative ponds.

Fig 6.10 Monthly mean DO concentrations in effluent from maturation ponds.
6.5 Temperature and DO Variations

As part of the evaluation of the WSP system performance various data on temperature and DO variations were collected for more than 1 year (Apr90-Jul92). Maximum and minimum temperature data were collected from mid-depth of each pond weekly for the period Apr90-Jan92. Data were also collected from the incoming sewage at the inlet works.

Temperature depth-time profiles were measured during the period Jan91-Jul92 in each pond. Additionally diurnal (24 hour cycle) data for DO and temperature were collected from each pond on 5 occasions during this period. The data collected above have been analysed in another research study therefore in this study discussions are of a general nature.

6.5.1 Maximum and Minimum Temperature Variations in Facultative and Maturation Ponds

Maximum, minimum and actual temperature of incoming sewage, facultative pond 1.1 and maturation pond 2.2 are shown in Figs 6.11, 6.12 and 6.13. The annual fluctuations in temperature were similar in all ponds and closely followed the expected seasonal variations.

![Graph showing temperature variations](image)

Fig 6.11 Weekly maximum, minimum and actual temperatures of raw sewage entering treatment works (Apr90-Jan92).
The weekly mean maximum temperature recorded for the raw sewage was 30.0°C with a high of 31.5°C recorded in Jul91 (summer season). The weekly mean minimum was 27.9°C with the lowest temperature of 24.5°C reported in Feb91 (winter season) (Fig 6.11).

![Weekly temperature graph for facultative pond 1.1 (Apr90-Jan92).](image)

**Fig 6.12 Weekly maximum, minimum and actual temperatures of facultative pond 1.1 (Apr90-Jan92).**

In facultative pond 1.1 the weekly mean maximum temperature noted was 30.6°C with a high of 33.0°C recorded in Aug90 (summer season). The weekly mean minimum was 27.5°C with the lowest temperature of 23.5°C reported in Dec90 (winter season) (Fig 6.12).

![Weekly temperature graph for maturation pond 2.2 (Apr90-Jan92).](image)

**Fig 6.13 Weekly maximum, minimum and actual temperatures of maturation pond 2.2 (Apr90-Jan92).**
The weekly mean maximum temperature reported for the final maturation pond was 30.7°C with the highest of 33.0°C recorded in May91 and Aug91 (summer season). The weekly mean minimum was 27.4°C with the lowest temperature of 23.0°C recorded in Dec90 and Feb91 (winter season) (Fig 6.13).

6.5.2 Temperature Profiles in Facultative and Maturation Ponds

Weekly measurements of temperature and DO concentrations at pond surface to a depth of 0.7 m in 0.1 m increments and at approximately 0900, 1200 and 1500 hours were carried out in the period (Jan91-Jul92). Measurements at the weir chamber (WC) represent pond liquid at a depth of 1.3 m and 1.0 m in the facultative pond 1.1 and maturation pond 2.2, respectively. For the purpose of presentation, the data have been summarised to demonstrate seasonal variations. The dry season (Dec-Apr) data are the means of 30 samples while data from the wet season (May-Nov) represent 43 samples. As the temperature distribution pattern was similar in both facultative ponds, only data from one facultative pond (1.1) are presented (Figs 6.14 and 6.15). In the maturation ponds, slightly higher temperatures were recorded in the final maturation pond 2.2 than in pond 2.1 probably due to the shallower depth. However as the patterns in both ponds were similar, temperature profile data from pond 2.2 were selected to be depicted (Figs 6.16 and 6.17).

For the majority of the research period all ponds were isothermal throughout the water column for the morning measurements. However in both seasons microstratification developed by noon and became more pronounced in the afternoon. Fluctuation in surface temperature was similar in all ponds and was generally higher than mean air temperature (Table 6.17). In the dry season temperature stratification of >2.0°C (difference between temperature measured at the surface and at 0.7 m) developed 4 times in the facultative ponds and on 11 occasions in the wet season. The final maturation pond 2.2 displayed temperature stratification 6 times in the dry season and on 23 occasions in the wet season.

At 0900 and at 1200 hours in the cool, dry season, water temperature in the facultative ponds between the surface and at a depth of 0.7 m varied <1°C and was thus considered thermally uniform. Whereas later in the afternoon (1500 hours), the surface temperature increased and was about 1.5°C higher than the temperature at 0.7 m (Fig 6.14). During the same period the maturation pond 2.2 the pattern was similar except that the difference between surface and 0.7 m was almost 2.0°C (Fig 6.16).
Fig 6.14  Mean temperature profile in facultative pond 1.1 in the dry season (Dec-Apr).

Fig 6.15  Mean temperature profile in facultative pond 1.1 in the wet season (May-Nov).

Fig 6.16  Mean temperature profile in maturation pond 2.2 in the wet season (May-Nov).

Fig 6.17  Mean temperature profile in maturation pond 2.2 in the wet season (May-Nov).
In the wet season the temperature profile became pronounced in both facultative and maturation ponds by noon with variations of almost 2.0°C from surface to 0.7 m depth. Later in the afternoon (1500 hours), the surface temperature increased and was approximately 3.0°C higher than at 0.7 m (Figs 6.16 and 6.17).

Although on average the ambient air temperature variation between the seasons is approximately 2.5°C, the night and day temperatures in either season may vary by more than 10°C. As daily stratification patterns are more important than seasonal changes in shallow bodies of water (Vaas and Sachlan, 1953; Young and Zimmerman, 1956; and Eriksen, 1966), observations of diurnal stratification in the ponds were made.

6.5.3 Diurnal Temperature Stratification in Facultative and Maturation Ponds

On 5 occasions measurement of temperature at various depths were carried out every 3 hours over a 24 hour cycle. In the dry season, measurements were made on 2 occasions in Jan91 and Mar91 while in the wet season 3 were done (May91, Jul91 and Sep91). Representing the dry and wet seasons (Jan91 and Sep91), data from the surface, at 0.3 m and 0.7 m depths of facultative pond 1.1 and maturation pond 2.2 are displayed in Figs 6.18-6.21.

In both ponds thermal stratification was observed on 10-11 Sep91 (Figs 6.19 and 6.21) but not significantly on 15-16 Jan91 (Figs 6.18 and 6.20). In Figs 6.19 and 6.21 it is shown that thermal stratification developed and was evident between 1200 to 2100 hours. By 2400 hours the stratification had broken down and ponds were isothermal.

During summer when incoming sewage has a lower temperature than the facultative ponds it will tend to stay on the bottom. In the winter when the sewage is warmer than the pond contents, surface streaming or sheet-spreading may occur (Marais, 1966). For that reason, consideration of thermal stratification has implications for the treatment efficiency because severe stratification will affect the sewage movement through the system (short-circuiting, retention time, removal efficiency).

The number of diurnal measurements limit the extent of discussion, however it may be assumed that in the Cayman ponds some stratification is inclined to occur diurnally. However, it is liable to be more significant on calm and sunny days as wind action will have the effect of breaking down stratification as will intense rainfall.
Fig 6.18 Pond 1.1 diurnal °C at the surface, 0.3 m and 0.7 m on 15-16 Jan91.

Fig 6.19 Pond 1.1 diurnal °C at the surface, 0.3 m and 0.7 m on 10-11 Sep91.

Fig 6.20 Pond 2.2 diurnal °C at the surface, 0.3 m and 0.7 m on 15-16 Jan91

Fig 6.21 Pond 2.2 diurnal °C at the surface, 0.3 m and 0.7 m on 10-11 Sep91.
The average temperature of incoming sewage in the summer is 29.5°C and 28.4°C in the dry/winter season. There is little difference between the average temperatures of the facultative ponds (average 29.0°C in summer and 27.0°C in winter) and the seasonal variations found in the raw sewage. Therefore, the streaming or sheet-spread effect caused by temperature differences between the sewage and the receiving water will not be pronounced compared with temperate and cold climates.

6.5.4 Dissolved Oxygen (DO) Profiles in Facultative and Maturation Ponds

Weekly measurements of DO concentrations at pond surface to a depth of 0.7 m in 0.1 m increments and at approximately 0900, 1200 and 1500 hours were carried out during the same period (Jan91-Jul92) as the temperature profiles. As in the preceding section the data have been summarised to demonstrate seasonal variations. The dry season (Dec-Apr) data are the means of 30 samples while data from the wet season (May-Nov) represent 43 samples. As the DO distribution profile was similar in both facultative ponds, only data from one facultative pond (1.1) are presented (Figs 6.22 and 6.23). There was a wider range in DO profile variations between the 2 maturation ponds, however it was decided to present only the data from the final maturation pond 2.2 (Figs 6.24 and 6.25).

DO concentrations in all ponds were <1.0 mg/l throughout the water column in the morning. However in both seasons DO stratification developed by noon and became more pronounced in the afternoon. In the hypolimnion (0.5 m and below), oxygen was rarely measured at concentrations above 1.0 mg/l in any pond. In the dry season DO stratification of >1.0 mg/l (difference between surface DO and at 0.7 m) developed 6 times (20% of weekly measurements) in the facultative ponds and on 21 occasions or 49% of the measurements in the wet season. The final maturation pond 2.2 exhibited DO stratification of >1.0 mg/l 100% of the time (30 times) in the dry season and 84% (36 occasions) in the wet season.

At 0900 and at 1200 hours in the dry season, DO in the facultative ponds between the surface and at a depth of 0.7 m varied little (0.5 mg/l) and was thus considered vertically homogenous. Whereas later in the afternoon (1500 hours), the oxygen levels at the surface to 0.3 m increased and were on average 2.0 mg/l higher than the DO at 0.7 m (Fig 6.22). During the same period the maturation pond 2.2 the pattern was similar except that the mean difference between surface and 0.7 m oxygen concentration was dramatically higher at 7.0 mg/l (Fig 6.24).
In the wet season, the vertical distribution of DO was pronounced in both facultative and maturation ponds by noon. Variations of up to 7.0 mg/l from surface to 0.7 m depth were observed (Fig 6.23 and 6.25).
Observations of diurnal DO stratification in the ponds were made in order to ascertain the extent of oxygen production through the depth of the lagoons.

6.5.5 Diurnal DO Stratification in Facultative and Maturation Ponds

Measurements of the DO at various depths were carried out every 3 hours over a 24 hour cycle on 5 occasions the same as previously reported for diurnal temperature profiles. Representing the dry and wet seasons (Jan91 and Sep91), DO data from the surface, at 0.3 m and 0.7 m depths of facultative pond 1.1 and maturation pond 2.2 are displayed in Figs 6.26-6.29. The aerators were not in use during the collection of this data.

In both ponds DO stratification was observed on 10-11 Sep91 (Figs 6.27 and 6.29) but not significantly on 15-16 Jan91 (Figs 6.26 and 6.28) when concentrations were in general <1.0 mg/l. However, maturation pond 2.2 did show increased DO (approximately 4.0 mg/l) by 1500 hours extending at least to 0.3 m (Fig 6.28). DO stratification developed and was more pronounced in the final maturation pond (Fig 6.29) between 1200 to 2100 hours. By 2400 hours the stratification had broken down and vertical distribution of DO was homogenous in all lagoons. Generally DO stratification coincided with thermal stratification reported and discussed in an earlier section (Section 6.5.3).

As with the diurnal temperature data, the number of measurements limit the depth of interpretation. However between 1200 and 2100 hours the maturation pond maintained DO concentrations >2.0 mg/l to at least a depth of 0.3 m (Fig 6.29). Further information on diurnal patterns and the extent of DO stratification will be useful in developing operational strategies (adjustments to the final effluent draw-off and the recirculation regime) to take advantage of the patterns.

Photosynthetic production, animal and plant respiration, bacterial respiration during decomposition of organic matter, chemical decomposition of dissolved organic matter and atmospheric gain all affect the oxygen balance in a body of water (Hutchinson, 1957). It is commonly accepted that photosynthesis by phytoplankton and macrophytes greatly affect oxygen concentration in maturation ponds (Mitchell and Williams, 1982).
Fig 6.26 Pond 1.1 diurnal DO at the surface, 0.3 m and 0.7 m on 15-16 Jan91.

Fig 6.27 Pond 1.1 diurnal DO at the surface, 0.3 m and 0.7 m on 10-11 Sep91.

Fig 6.28 Pond 2.2 diurnal DO at the surface, 0.3 m and 0.7 m on 15-16 Jan91.

Fig 6.29 Pond 2.2 diurnal DO at the surface, 0.3 m and 0.7 m on 10-11 Sep91.
6.6 Suspended, Volatile and Fixed Solids Variations

Analysis of suspended solids (SS) was expanded during this period (Apr90-Mar93) to include volatile (VSS) and fixed solids (FSS). The data are summarised in Tables 6.11-6.14.

As shown in Table 6.11 there was little difference in the mean SS of the raw sewage and pond effluents. On average there was 10% reduction in SS at the facultative stage in contrast to the maturation stage where generally the SS increased. Removal efficiency in the WSP was poor during this period with an overall removal of only 1%.

Table 6.11 Suspended solids removal in facultative and maturation ponds Apr90-Mar93.

<table>
<thead>
<tr>
<th>Suspended solids mg/l</th>
<th>Mean</th>
<th>Max</th>
<th>Min</th>
<th>% removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>IN SEW</td>
<td>128</td>
<td>246</td>
<td>59</td>
<td>n/a</td>
</tr>
<tr>
<td>PD 1.1</td>
<td>117</td>
<td>227</td>
<td>59</td>
<td>9</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>116</td>
<td>207</td>
<td>55</td>
<td>10</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>132</td>
<td>269</td>
<td>77</td>
<td>-13</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>127</td>
<td>221</td>
<td>70</td>
<td>4</td>
</tr>
<tr>
<td>overall % removal</td>
<td></td>
<td></td>
<td></td>
<td>1</td>
</tr>
</tbody>
</table>

The VSS data reported in Table 6.12 is an indication of combustible organic matter in the raw sewage and pond effluents. Throughout the WSP system reduction was observed in the facultative ponds however in the first maturation pond a significant increase (17%) in VSS was found. This reflects the increase expected at the first maturation stage due to the development and growth of biomass (algae). Throughout the WSP a mean increase of 6% was seen.

Table 6.12 Volatile suspended solids (VSS) removal in facultative and maturation ponds Apr90-Mar93.

<table>
<thead>
<tr>
<th>Volatile suspended solids mg/l</th>
<th>Mean</th>
<th>Max</th>
<th>Min</th>
<th>% removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>IN SEW</td>
<td>84</td>
<td>125</td>
<td>47</td>
<td>n/a</td>
</tr>
<tr>
<td>PD 1.1</td>
<td>79</td>
<td>116</td>
<td>41</td>
<td>6</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>75</td>
<td>117</td>
<td>23</td>
<td>11</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>90</td>
<td>121</td>
<td>50</td>
<td>-17</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>89</td>
<td>126</td>
<td>49</td>
<td>1</td>
</tr>
<tr>
<td>overall % removal</td>
<td></td>
<td></td>
<td></td>
<td>-6</td>
</tr>
</tbody>
</table>
The FSS data summary presented in Table 6.13 represents that part of the total SS that is not volatile. The mean FSS of the raw sewage and that of the final effluent did not change appreciably overall. However the pattern of FSS removal was similar as with the VSS where the first maturation pond 2.1 exhibited an increase in concentrations compared to the remainder of the system.

Table 6.13 Fixed suspended solids (FSS) removal in facultative and maturation ponds Apr90-Mar93.

<table>
<thead>
<tr>
<th></th>
<th>Mean</th>
<th>Max</th>
<th>Min</th>
<th>% removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>IN SEW</td>
<td>36</td>
<td>65</td>
<td>18</td>
<td>n/a</td>
</tr>
<tr>
<td>PD 1.1</td>
<td>36</td>
<td>88</td>
<td>17</td>
<td>0</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>35</td>
<td>79</td>
<td>0</td>
<td>5</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>39</td>
<td>74</td>
<td>13</td>
<td>-12</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>36</td>
<td>84</td>
<td>0</td>
<td>8</td>
</tr>
<tr>
<td>overall % removal</td>
<td></td>
<td></td>
<td></td>
<td>0</td>
</tr>
</tbody>
</table>

In Table 6.14, a comparison of the percentage of the volatile and fixed solids that comprised the SS is shown. On average, the total SS of the raw sewage consisted of 65% VSS and 28% FSS. The VSS increased to represent 70% of the total SS in the final effluent while the FSS showed only a slight increase to 29%. The percentage of VSS in the Cayman WSP is lower than that typically reported in pond effluents due to reduced algal biomass. Elsewhere VSS may be as high as of 80% (Pearson and Silva, 1989).

Table 6.14 Volatile suspended solids and fixed suspended solids as a percentage of total suspended solids during the period Apr90-Mar93.

<table>
<thead>
<tr>
<th>VSS and FSS as a % of the total SS</th>
<th>VSS</th>
<th>FSS</th>
</tr>
</thead>
<tbody>
<tr>
<td>IN SEW</td>
<td>65</td>
<td>28</td>
</tr>
<tr>
<td>PD 1.1</td>
<td>67</td>
<td>31</td>
</tr>
<tr>
<td>PD 1.2</td>
<td>64</td>
<td>30</td>
</tr>
<tr>
<td>PD 2.1</td>
<td>68</td>
<td>30</td>
</tr>
<tr>
<td>PD 2.2</td>
<td>70</td>
<td>29</td>
</tr>
</tbody>
</table>
Minimal change in the composition of suspended solids in the sewage and throughout the ponds system was observed during monitoring phase II (Table 6.14).

6.7 Sludge Depth Measurements

As wastewater slowly passes through a WSP system, a high proportion of the suspended solids in the raw sewage is deposited (including parasitic ova, bacteria, grit, etc.) in the ponds. Anaerobic decomposition, concentration and some mineralisation are several of the processes that occur in the accumulated sludge. Most of the SS in facultative ponds settle to the bottom developing a layer that works as an anaerobic sludge digestor (Ellis, 1983). Generally the main activities taking place in the sludge layer are acid fermentation and methanogenesis, however in the Cayman WSP system these processes are influenced by the high sulphate concentrations in the wastewater leading to the excessive production of hydrogen sulphide (discussed in Chapter 8).

One of the benefits of a WSP system is that the rate of sludge accumulation is slow. A wide range of sludge accumulation rates in WSP have been reported in literature. Mara (1976) reports that desludging of facultative ponds is infrequently required, once every 10-15 years when a thickness of 25.0 cm or less is expected. Middlebrooks et al (1982) reported, from a limited investigation of sludge accumulation in experimental ponds (in Fayette, Missouri) receiving municipal waste, that the most significant factor in sludge deposition in WSP is the silt and other inorganic material entering the lagoon. They reported that in 15 facultative ponds ranging in age from 5 months to 82 months the maximum mean sludge accumulation was 17.4 cm. Schetrite and Racault (in press) reported sludge accumulation of 16.5 cm in the first pond of a 3 pond WSP system in France after 11 years of operation. However Saqqar and Pescod (in press b) reported sludge accumulation of 1.7 m in an anaerobic pond in Alsamra, Jordan after only 44 months of operation. Gloyna (1971) suggests for design purposes an accumulation rate of 0.03 m³/capita/year be used. In India, Arceivala et al (1970) reported sludge accumulation rates ranging 0.08-0.30 m³/capita/year.

Monitoring of sludge deposition is important for operational purposes as it affects the pond volume available for treatment, short-circuiting and retention time. Measurement of sludge accumulation in the Cayman WSP system commenced on a biannual basis in Apr90 using the method described by Pearson et al (1987a). The method recommends a minimum of 5 points be measured in each pond and then used to calculate the mean depth. In the Cayman facultative ponds, 16 points were identified along the aerator
suspension lines. In the maturation ponds, 12 points were measured at specific points. Measurements were taken from a boat (Fig 4.12, pg 77).

Table 6.15 presents a summary of annual average sludge depth in each pond:

<table>
<thead>
<tr>
<th>DATE</th>
<th>PD 1.1</th>
<th>PD 1.2</th>
<th>PD 2.1</th>
<th>PD 2.2</th>
</tr>
</thead>
<tbody>
<tr>
<td>1990</td>
<td>0.145</td>
<td>0.164</td>
<td>0.054</td>
<td>0.041</td>
</tr>
<tr>
<td>1991</td>
<td>0.346</td>
<td>0.294</td>
<td>0.215</td>
<td>0.241</td>
</tr>
<tr>
<td>1992</td>
<td>0.385</td>
<td>0.362</td>
<td>0.177</td>
<td>0.217</td>
</tr>
</tbody>
</table>

Middlebrooks et al (1982), in their analysis of sludge accumulation and distribution in facultative ponds, concluded that the direction of prevailing wind did not significantly affect the pattern of sludge deposition. This does not appear to be the case in the Cayman WSP system where the prevailing wind direction is easterly and the deepest sludge pockets are found in the westernmost corners of all ponds (Figs 7.5-7.12). This indicates that sludge build-up is perpendicular to and directly affected by the prevailing eastern wind. Middlebrooks et al (1982) theorised that the greater accumulation of sludge in the corners is probably due to sludge accumulation around the inlet.

6.8 Septage Quality and Flow
Monitoring of the quality of septage received at the Authority's treatment works commenced in Jan91. Samples of 11 volumes are from collected from each tanker and composited over a period of 1 day. Various chemical and biological parameters are routinely analysed as shown in the annual summaries for monitoring phase II in Table 6.16. Results for most parameters vary widely in septage samples. The chemical characteristics of the septage in Cayman is typical where reported variations in BODuf may range from 970-25000 mg/l, CODuf as high as 80600 mg/l, and SS up to 110000 mg/l (Metcalf and Eddy, 1972; and Polprasert et al, 1984). The volume of septage is <1% of the average daily flow to the treatment works therefore its influence on performance is minimal.
Table 6.16  Annual summary of composite septage samples analysed in 1991 and 1992.

<table>
<thead>
<tr>
<th></th>
<th>BODuf mg/l</th>
<th>CODuf mg/l</th>
<th>EC μS/cm</th>
<th>SO₄ mg/l</th>
<th>NH₃-N mg/l</th>
<th>SS mg/l</th>
<th>VSS mg/l</th>
<th>FSS mg/l</th>
<th>pH</th>
<th>Fec coli cfu/100 ml</th>
</tr>
</thead>
<tbody>
<tr>
<td>1991</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mean</td>
<td>871.3</td>
<td>4065.2</td>
<td>4883.7</td>
<td>469.7</td>
<td>207.7</td>
<td>3606.6</td>
<td>3071.3</td>
<td>576.5</td>
<td>7.41</td>
<td>1.76 x 10⁶</td>
</tr>
<tr>
<td>Min</td>
<td>180.2</td>
<td>398.1</td>
<td>1038.0</td>
<td>18.0</td>
<td>4.0</td>
<td>128.2</td>
<td>100.0</td>
<td>99.5</td>
<td>4.72</td>
<td>1.00 x 10⁶</td>
</tr>
<tr>
<td>Max</td>
<td>2400.0</td>
<td>16177.3</td>
<td>18640.0</td>
<td>2288.0</td>
<td>1680.0</td>
<td>23076.9</td>
<td>22115.4</td>
<td>3666.7</td>
<td>9.17</td>
<td>1.68 x 10⁷</td>
</tr>
<tr>
<td>No samples</td>
<td>39</td>
<td>38</td>
<td>41</td>
<td>21</td>
<td>39</td>
<td>39</td>
<td>35</td>
<td>35</td>
<td>40</td>
<td>27</td>
</tr>
<tr>
<td>1992</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mean</td>
<td>789.5</td>
<td>5716.6</td>
<td>4050.1</td>
<td>186.1</td>
<td>229.5</td>
<td>3069.0</td>
<td>2992.7</td>
<td>545.1</td>
<td>7.43</td>
<td>4.61 x 10⁶</td>
</tr>
<tr>
<td>Min</td>
<td>180.2</td>
<td>1443.4</td>
<td>1100.0</td>
<td>18.0</td>
<td>32.0</td>
<td>160.0</td>
<td>125.0</td>
<td>0.0</td>
<td>5.00</td>
<td>4.00 x 10⁶</td>
</tr>
<tr>
<td>Max</td>
<td>1440.1</td>
<td>17289.5</td>
<td>11910.0</td>
<td>1260.0</td>
<td>1600.0</td>
<td>33000.0</td>
<td>29416.5</td>
<td>3583.3</td>
<td>9.31</td>
<td>3.03 x 10⁷</td>
</tr>
<tr>
<td>No samples</td>
<td>18</td>
<td>14</td>
<td>22</td>
<td>12</td>
<td>17</td>
<td>21</td>
<td>18</td>
<td>18</td>
<td>22</td>
<td>21</td>
</tr>
</tbody>
</table>

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6.9 Meteorological Data 1990-1992

Collection of meteorological data continued during monitoring phase II and are summarised in Table 6.17.


<table>
<thead>
<tr>
<th>METEOROLOGICAL DATA</th>
<th>1990</th>
<th>1991</th>
<th>1992</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rainfall in mm</td>
<td>Met</td>
<td>STW</td>
<td>Met</td>
</tr>
<tr>
<td>total rainfall</td>
<td>1372.8</td>
<td>1729.1</td>
<td>1510.4</td>
</tr>
<tr>
<td>annual daily average</td>
<td>3.76</td>
<td>4.70</td>
<td>4.14</td>
</tr>
<tr>
<td>daily average, dry season (dec-apr)</td>
<td>1.92</td>
<td>2.90</td>
<td>0.94</td>
</tr>
<tr>
<td>daily average, wet season (may-nov)</td>
<td>5.06</td>
<td>6.10</td>
<td>6.39</td>
</tr>
<tr>
<td>Wind direction in degrees ° from north</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>annual average</td>
<td>81.7</td>
<td>86.7</td>
<td>81.7</td>
</tr>
<tr>
<td>average, dry season (dec-apr)</td>
<td>82.0</td>
<td>90.0</td>
<td>88.0</td>
</tr>
<tr>
<td>average, wet season (may-nov)</td>
<td>81.4</td>
<td>84.3</td>
<td>77.1</td>
</tr>
<tr>
<td>Windspeed m/sec</td>
<td>Met</td>
<td>STW</td>
<td>Met</td>
</tr>
<tr>
<td>average</td>
<td>3.5</td>
<td>2.5</td>
<td>3.9</td>
</tr>
<tr>
<td>average, dry season (dec-apr)</td>
<td>3.9</td>
<td>4.4</td>
<td>2.8</td>
</tr>
<tr>
<td>average, wet season (may-nov)</td>
<td>3.2</td>
<td>3.5</td>
<td>3.5</td>
</tr>
<tr>
<td>Ambient air temperature °C</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>average</td>
<td>27.8</td>
<td>27.6</td>
<td>27.7</td>
</tr>
<tr>
<td>dry season (dec-apr)</td>
<td>26.4</td>
<td>26.5</td>
<td>26.4</td>
</tr>
<tr>
<td>wet season (may-nov)</td>
<td>28.8</td>
<td>28.4</td>
<td>28.5</td>
</tr>
</tbody>
</table>

Note: Met = data from Civil Aviation Meteorological Station; STW = data from sewage treatment works.

Rainfall data were collected at the sewage treatment works as was the windspeed. There are variations between data from the treatment works and from the Civil Aviation Meteorological Station. This may be attributed to the differences in conditions at the two locations. In general, rainfall measured at the sewage treatment works was less than that from the Meteorological Station as were windspeed data. This may be attributed to vegetation and other obstacles on the windward side of the sewage treatment works, and to the height of the anemometer related to the ground level. In contrast, the Meteorological Station is located at the airport where data collection takes place in a wide open area unobstructed by structures or vegetation.

6.10 Summary of Monitoring Phase II

Generally, the WSP system demonstrated reduced performance attributable to hydraulic overloading and the effect of high sulphate concentrations in the incoming wastewater flow. The following observations are made on the results of this monitoring phase:
1. The extent of groundwater intrusion into the sewerage system increased significantly and was evidenced clearly by the increasing salinity and flow to the sewage works.

2. Flow to the treatment works exceeded the 1996 design flows for more than 16 months during this period. Consequently, the WSP system was hydraulically overloaded, significantly reducing the nominal retention time by 65% to 14 days.

3. By the end of Jan93 salinity of the incoming wastewater was approaching 25000 μS/cm, more than double the salinity at the end of monitoring phase I (Chapter 5). At this stage, more than 40% of the incoming flow was attributed to groundwater intrusion.

4. The dramatic effect of repair of major cracks in the sewers was clearly seen in the reduction of incoming flow from an average of approximately 5000 m³/day in Feb93 to 3000 m³/day in Mar93 when the first phase of the sewer rehabilitation was completed.

5. As the raw sewage BODuf was increasingly influenced by the diluting effect of the infiltrating groundwater, overall percentage BODuf removal dropped significantly from 87% in phase I to 73% in phase II.

6. CODuf loading increased dramatically in this phase reflecting the influence of inorganic oxygen demand due to high sulphide production in the ponds. Accordingly overall removal efficiency decreased from the previous monitoring phase, from 55% to 32%.

7. Overall faecal coliforms removal in this period achieved a 3 log reduction. The final effluent was unable to meet reuse guidelines for irrigation due to unacceptable levels of faecal coliforms and high salinity.

8. Temperature profiles indicated that the ponds were isothermal in the morning and minor stratification developed by noon and was more pronounced in the shallower maturation ponds (>3.0°C) in the wet season when temperatures are higher. Dissolved oxygen distribution in the facultative ponds did not increase significantly in the dry season with concentrations at the surface <1.5 mg/l. The maturation ponds however displayed significant stratification by
mid-afternoon in both the dry and wet season. It is significant that the effluent drawoff level in the final maturation pond is approximately 1.0 m deep and at this depth the oxygen concentration is $<1.0$ mg/l. This is an important consideration in evaluating the usefulness of recirculating the effluent from this pond.

9. Sludge depth in all ponds increased in 1991 and 1992, reducing effective pond volume and retention times. By the end of the monitoring period, based on design pond volumes and average sludge depths, the effective volume of each facultative pond was reduced by 18-19% while that of each maturation ponds was reduced by 14%.

10. In the following chapter a general review of the effect of reduced flow and decreased salinity on the performance of the WSP system is undertaken as well as an overview of the system’s performance since commissioning in 1988.
CHAPTER 7

7.0 SUMMARY & ROUTINE MONITORING APRIL 1993 TO DECEMBER 1994

7.1 Monitoring of WSP Apr93 - Dec94

In this Chapter it is intended to present a summary of the entire monitoring programme which spanned the 7 years of the Cayman Island's WSP operation. There are a number of activities and important observations during the course of 1993-1994, for instance:

1. In situ repair, consisting primarily of 3.7 km of 150 mm vitrified clay pipes commenced in late 1992, was completed in Mar/Apr93 using a ‘trenchless’ pipe rehabilitation method known as U-Liner. The repairs resulted in 40% reduction of flow to the treatment works and electricity costs dropped by 22%.

2. Based on the assumption that the reduction in saline groundwater intrusion (both in salinity and in volume) effected by the repairs would reduce odour problems at the sewage treatment works, all aerators were removed from the facultative ponds in Mar93.

3. An intensive investigation of the structural conditions of manholes was carried out in 1993 which revealed that 38 manholes were in poor condition and required structural repairs in order to prevent further deterioration of the concrete. Repairs are scheduled to commence in 1995.

4. A close circuit TV (CCTV) survey of the sewers was carried out on the remaining unrepaired sewers. The survey revealed that more than 40% of the 150 mm and 100 mm clay pipes were severely cracked or damaged. The majority of the damaged pipes were under road-crossings. The WAC plans to repair about 600 m of 150 mm and 100 mm pipes in Jan95.

5. The reduction in flow to the works was accompanied by a marked decrease in the salinity of the raw sewage as seen in Fig 7.1.

6. A tracer study was carried out in Sep94 to evaluate the hydraulic efficiency of the sewage treatment works (this study is described and discussed in Chapter 10).
7. Sampling frequency was reduced to twice per month due mainly to the author's expanding duties and responsibilities in other areas of the Water Authority's work.

The repairs carried out on the system reduced the salinity and had a profound effect on the system. The data presented and summarised in this Chapter will be mainly in the form of annual averages. Although the system was subjected to less saline sewage, performance did not 'overwhelmingly improve' but did however show improved faecal coliform and BOD removals.

7.2 Decreasing Flows and Reduced Salinity

Fig 7.1 clearly summarises the rise and fall of sewage flow and salinity during the entire study and highlights the 3 year period (1990-1992) when flow exceeded the design.

Immediately after the repairs were completed in 1993, the salinity of the raw sewage reduced from approaching the 25000 µS/cm levels to <15000 µS/cm. However by the end of 1993 salinity began to increase again and in 1994 varied between 10000-17000 µS/cm. The mean values for each pond are shown in Table 7.1:

<table>
<thead>
<tr>
<th>Year</th>
<th>Mean Daily Flow</th>
<th>IN SEW</th>
<th>PD 1.1</th>
<th>PD 1.2</th>
<th>PD 2.1</th>
<th>PD 2.2</th>
<th>SEPTAGE</th>
</tr>
</thead>
<tbody>
<tr>
<td>1988</td>
<td>741</td>
<td>3787</td>
<td>3006</td>
<td>3257</td>
<td>3057</td>
<td>3621</td>
<td>n/d</td>
</tr>
<tr>
<td>1989</td>
<td>1599</td>
<td>6551</td>
<td>6600</td>
<td>6560</td>
<td>6588</td>
<td>6580</td>
<td>n/d</td>
</tr>
<tr>
<td>1990</td>
<td>OVER LOAD-2852</td>
<td>11955</td>
<td>11440</td>
<td>11697</td>
<td>11936</td>
<td>11717</td>
<td>n/d</td>
</tr>
<tr>
<td>1991</td>
<td>OVERLOAD -4827</td>
<td>16749</td>
<td>16980</td>
<td>16674</td>
<td>16477</td>
<td>16373</td>
<td>4884</td>
</tr>
<tr>
<td>1992</td>
<td>OVERLOAD-4748</td>
<td>21282</td>
<td>22221</td>
<td>22364</td>
<td>22187</td>
<td>22242</td>
<td>4050</td>
</tr>
<tr>
<td>1993</td>
<td>2952</td>
<td>17462</td>
<td>17088</td>
<td>17249</td>
<td>17321</td>
<td>17468</td>
<td>4644</td>
</tr>
<tr>
<td>1994</td>
<td>2933</td>
<td>13303</td>
<td>13702</td>
<td>13646</td>
<td>13489</td>
<td>13483</td>
<td>5338</td>
</tr>
</tbody>
</table>

Table 7.1 Annual electrical conductivity (µS/cm) in the Cayman Islands' waste stabilisation ponds, raw sewage and septage, and the annual mean of daily incoming sewage flow (m³/day) to the treatment works (1988-1994).
7.2.1 Faecal Coliform Indicator Removal

In Fig 7.2 the faecal coliform data from the commencement of the study to 1994 are plotted. It is clear from the graph that achieving even the WHO/Engleberg reuse guidelines (<1000 fec/100 ml) was impossible under the operating conditions of the ponds.

The annual (mean) faecal coliform removal efficiency for each pond is shown in Table 7.2. Considering faecal coliform removal over the entire monitoring period (1988-94) it appears that the maturation ponds are much more vulnerable to damage by hydraulic overload and salinity than the facultative ponds.

Comparing Table 7.2 and Figs 7.1-7.2, whereas in 1990, the first year that the maximum design flow was exceeded, both facultative ponds held to >99% removal performance, the second maturation pond (2.2) dropped 12% (from 84%-72%). By 1992, the second maturation pond had dropped a further 8%, to 64%. Even in 1993, following sewer repair, with flows back below the design maximum, the maturation ponds did not recover to the 1990 performance levels.

It can therefore be deduced that excess flow is not the primary problem in the maturation ponds but rather the possible combination of H₂S carry-over and short-circuiting, in spite of (or perhaps partly because of) the baffles added in 1990. The problem of the effect of H₂S on the ecology is therefore investigated in the first section of Chapter 9 and short-circuiting in Chapter 10.
Table 7.2 Summary of faecal coliform percentage removal for the sewage treatment works (1988-1994).

<table>
<thead>
<tr>
<th>Year</th>
<th>FAC PD 1.1 % removal</th>
<th>FAC PD 1.2 % removal</th>
<th>MAT PD 2.1 % removal</th>
<th>MAT PD 2.2 % removal</th>
<th>OVERALL % removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>1988</td>
<td>98.078</td>
<td>99.291</td>
<td>90.251</td>
<td>68.333</td>
<td>99.962</td>
</tr>
<tr>
<td>1989</td>
<td>99.876</td>
<td>99.490</td>
<td>78.009</td>
<td>84.554</td>
<td>99.998</td>
</tr>
<tr>
<td>1990</td>
<td>99.953</td>
<td>99.371</td>
<td>82.720</td>
<td>72.318</td>
<td>99.998</td>
</tr>
<tr>
<td>1991</td>
<td>93.724</td>
<td>95.194</td>
<td>84.722</td>
<td>64.049</td>
<td>99.440</td>
</tr>
<tr>
<td>1992</td>
<td>94.845</td>
<td>93.188</td>
<td>85.219</td>
<td>67.117</td>
<td>99.616</td>
</tr>
<tr>
<td>1993</td>
<td>96.281</td>
<td>96.818</td>
<td>77.281</td>
<td>71.747</td>
<td>99.868</td>
</tr>
<tr>
<td>1994</td>
<td>98.026</td>
<td>98.274</td>
<td>81.036</td>
<td>71.665</td>
<td>99.904</td>
</tr>
</tbody>
</table>
Fig 7.2 Faecal coliform densities in raw sewage and all pond effluents analysed 1988-1994 compared with the reuse guidelines from WHO and California.
The effects of hydraulic overload on the system is clearly shown in Table 7.3 where the theoretical retention time for each pond and the entire system is presented. As the daily inflow increased, retention time reduced and thus the ability to efficiently remove the faecal indicator bacteria was reduced.

Table 7.3 Summary of theoretical retention time (days) in the Cayman waste stabilisation ponds (1988-1994).

<table>
<thead>
<tr>
<th>Year</th>
<th>PD 1.1</th>
<th>PD 1.2</th>
<th>PD 2.1</th>
<th>PD 2.2</th>
<th>TOTAL RET</th>
</tr>
</thead>
<tbody>
<tr>
<td>1988</td>
<td>22.9</td>
<td>22.9</td>
<td>16.2</td>
<td>13.5</td>
<td>75.6</td>
</tr>
<tr>
<td>1989</td>
<td>10.6</td>
<td>10.6</td>
<td>7.5</td>
<td>6.3</td>
<td>35.0</td>
</tr>
<tr>
<td>1990</td>
<td>6.1</td>
<td>5.9</td>
<td>4.2</td>
<td>3.7</td>
<td>19.8</td>
</tr>
<tr>
<td>1991</td>
<td>2.9</td>
<td>3.0</td>
<td>2.2</td>
<td>1.9</td>
<td>10.0</td>
</tr>
<tr>
<td>1992</td>
<td>3.3</td>
<td>3.5</td>
<td>2.5</td>
<td>2.1</td>
<td>11.3</td>
</tr>
<tr>
<td>1993</td>
<td>4.9</td>
<td>5.0</td>
<td>3.6</td>
<td>2.9</td>
<td>16.3</td>
</tr>
<tr>
<td>1994</td>
<td>4.9</td>
<td>4.9</td>
<td>3.5</td>
<td>2.7</td>
<td>16.1</td>
</tr>
</tbody>
</table>

Note: calculations for 1988 & 1989 are based on design pond volumes, however data for remaining years are calculated based on pond volumes calculations.

7.2.2 BOD and COD Removals

The BODuf concentration variations in the raw sewage and effluent from the sewage ponds over the period 1988-1994 are shown on Fig 7.3.

It is clear from Fig 7.3 that even with the reduction in removal efficiency through the period 1988-1994 the final effluent from maturation pond 2.2 was able to meet the WA own discharge regulations of ≤30 mg/l.

For overall BODuf removal, the year-on-year trend is presented in Table 7.4 and shows progressively deteriorating performance from 1988 to 1992. In 1993, with major sewer rehabilitation completed, and a return to flows below the 1996 design maximum, overall BODuf removal recovers from 65% (1992) to 74%, with a further recovery to 78% in 1994. This is in spite of flow rising again above the design flow for part of that year. As expected, the majority of the removal takes place in the facultative ponds. Comparing the relative contribution of the two maturation ponds for BODuf removal, again as with faecal coliform removal, it is the second maturation pond (2.2) which has
the poorest performance. In 1993, this maturation pond, contributes nothing (0%) whilst for 3 of the last 4 years it actually increases BODuf values!

Table 7.4 Summary of BODuf percentage removal for the WSP system (1988-94).

<table>
<thead>
<tr>
<th>Year</th>
<th>Facultative pond 1.1</th>
<th>Facultative pond 1.2</th>
<th>Maturation pond 2.1</th>
<th>Maturation pond 2.2</th>
<th>Overall performance</th>
</tr>
</thead>
<tbody>
<tr>
<td>1988</td>
<td>81</td>
<td>85</td>
<td>38</td>
<td>-20</td>
<td>88</td>
</tr>
<tr>
<td>1989</td>
<td>71</td>
<td>72</td>
<td>9</td>
<td>45</td>
<td>86</td>
</tr>
<tr>
<td>1990 Overload</td>
<td>62</td>
<td>63</td>
<td>22</td>
<td>16</td>
<td>76</td>
</tr>
<tr>
<td>1991 Overload</td>
<td>59</td>
<td>56</td>
<td>29</td>
<td>-10</td>
<td>67</td>
</tr>
<tr>
<td>1992 Overload</td>
<td>61</td>
<td>54</td>
<td>34</td>
<td>-25</td>
<td>65</td>
</tr>
<tr>
<td>1993</td>
<td>62</td>
<td>68</td>
<td>25</td>
<td>0</td>
<td>74</td>
</tr>
<tr>
<td>1994</td>
<td>68</td>
<td>70</td>
<td>34</td>
<td>-4</td>
<td>78</td>
</tr>
</tbody>
</table>

*Note: negative values indicate increases rather than removal.*

The performance deterioration in terms of BOD and faecal coliform percentage removals has been demonstrated over a period of 7 years in this sewage treatment system. This reduction has been attributed mainly to circumstances involving the breakdown of the sewerage system and the increasing intrusion of saline groundwater as reported previously by Frederick (1991a, 1991b).

From an operational and reuse point this has created a difficult situation involving costly relining and rehabilitation of the clay sewers, manholes and pumping stations in 1993 as was indicated in Chapter 3, 5 and 6.

The BODuf surface loading trend throughout the system is shown in Table 7.5.

The annual average concentrations of CODuf and removal efficiency through the ponds since 1988-1994 are shown in Tables 7.6 and 7.7.
Fig 7.3 Average monthly BOD₅ of incoming raw sewage and all pond effluents (1988-1994) compared to WA Discharge Regulations for Effluent BOD.
Table 7.5 Summary of BODuf surface loading in kg/ha day on each pond at the WSP for the period 1988-1994.

<table>
<thead>
<tr>
<th>YEAR</th>
<th>FAC PD 1.1</th>
<th>FAC PD 1.2</th>
<th>MAT PD 2.1</th>
<th>MAT PD 2.2</th>
</tr>
</thead>
<tbody>
<tr>
<td>1988</td>
<td>72</td>
<td>72</td>
<td>16</td>
<td>11</td>
</tr>
<tr>
<td>1989</td>
<td>163</td>
<td>163</td>
<td>46</td>
<td>51</td>
</tr>
<tr>
<td>1990</td>
<td>162</td>
<td>163</td>
<td>100</td>
<td>64</td>
</tr>
<tr>
<td>1991</td>
<td>162</td>
<td>163</td>
<td>110</td>
<td>71</td>
</tr>
<tr>
<td>1992</td>
<td>165</td>
<td>165</td>
<td>76</td>
<td>74</td>
</tr>
<tr>
<td>1993</td>
<td>174</td>
<td>174</td>
<td>61</td>
<td>46</td>
</tr>
<tr>
<td>1994</td>
<td>198</td>
<td>200</td>
<td>83</td>
<td>56</td>
</tr>
</tbody>
</table>
Table 7.6 Mean annual CODuf in mg/l and percentage reductions in facultative ponds in the period 1988-1994.

<table>
<thead>
<tr>
<th>Year</th>
<th>IN SEW</th>
<th>PD 1.1</th>
<th>% removal</th>
<th>PD 1.2</th>
<th>% removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>1988</td>
<td>284</td>
<td>231</td>
<td>18</td>
<td>129</td>
<td>55</td>
</tr>
<tr>
<td>1989</td>
<td>399</td>
<td>208</td>
<td>48</td>
<td>213</td>
<td>47</td>
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<tr>
<td>1990</td>
<td>524</td>
<td>386</td>
<td>26</td>
<td>373</td>
<td>29</td>
</tr>
<tr>
<td>1991</td>
<td>461</td>
<td>388</td>
<td>16</td>
<td>395</td>
<td>14</td>
</tr>
<tr>
<td>1992</td>
<td>539</td>
<td>293</td>
<td>46</td>
<td>291</td>
<td>46</td>
</tr>
<tr>
<td>1993</td>
<td>406</td>
<td>292</td>
<td>28</td>
<td>286</td>
<td>30</td>
</tr>
<tr>
<td>1994</td>
<td>385</td>
<td>310</td>
<td>19</td>
<td>306</td>
<td>21</td>
</tr>
</tbody>
</table>
Table 7.7 Mean annual CODuf in mg/l and percentage reductions in maturation ponds during the period 1988-1994.

<table>
<thead>
<tr>
<th>Year</th>
<th>PD 2.1</th>
<th>% removal</th>
<th>PD 2.2</th>
<th>% removal</th>
<th>% removal overall system</th>
</tr>
</thead>
<tbody>
<tr>
<td>1988</td>
<td>137</td>
<td>24</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>1989</td>
<td>197</td>
<td>6</td>
<td>183</td>
<td>7</td>
<td>54</td>
</tr>
<tr>
<td>1990</td>
<td>330</td>
<td>13</td>
<td>328</td>
<td>1</td>
<td>37</td>
</tr>
<tr>
<td>1991</td>
<td>360</td>
<td>8</td>
<td>354</td>
<td>2</td>
<td>23</td>
</tr>
<tr>
<td>1992</td>
<td>282</td>
<td>3</td>
<td>273</td>
<td>3</td>
<td>49</td>
</tr>
<tr>
<td>1993</td>
<td>311</td>
<td>-8</td>
<td>277</td>
<td>11</td>
<td>32</td>
</tr>
<tr>
<td>1994</td>
<td>273</td>
<td>-11</td>
<td>273</td>
<td>0</td>
<td>29</td>
</tr>
</tbody>
</table>
During the first 4 years of data the CODuf % removal efficiency in the WSP system reduced throughout the ponds as the salinity and flow increased. Overall removal efficiency however improved in 1992 although the salinity and flow were both at their peak. Subsequently since the reduction of both flow and salinity in 1993, the overall removal efficiency has decreased.

7.3 Summary of Other Parameters Monitored (1988-1994)
In this section general summaries of other monitored parameters: ammonia-nitrogen, sulphate, sulphide, pH and temperature are presented in tabular format.

7.3.1 Ammonia-nitrogen (NH₃-N) in WSP 1988-1994
The annual average concentrations of NH₃-N and removal efficiency through the ponds for the period 1988-1994 are shown in Tables 7.8 and 7.9.
Table 7.8  Annual average NH$_3$-N concentrations in mg/l and percentage reductions in facultative ponds (1988-1994).

<table>
<thead>
<tr>
<th>Year</th>
<th>IN SEW</th>
<th>PD 1.1 removal</th>
<th>PD 1.2 removal</th>
</tr>
</thead>
<tbody>
<tr>
<td>1988</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>1989</td>
<td>51.3</td>
<td>69</td>
<td>14.8</td>
</tr>
<tr>
<td>1990</td>
<td>35.4</td>
<td>53</td>
<td>71</td>
</tr>
<tr>
<td>1991</td>
<td>38.5</td>
<td>68</td>
<td>56</td>
</tr>
<tr>
<td>1992</td>
<td>17.0</td>
<td>43</td>
<td>67</td>
</tr>
<tr>
<td>1993</td>
<td>32.0</td>
<td>18.6</td>
<td>47</td>
</tr>
<tr>
<td>1994</td>
<td>29.9</td>
<td>14.0</td>
<td>13.3</td>
</tr>
<tr>
<td>Year</td>
<td>PD 2.1 removal</td>
<td>PD 2.2 removal</td>
<td>overall system removal</td>
</tr>
<tr>
<td>------</td>
<td>----------------</td>
<td>----------------</td>
<td>-----------------------</td>
</tr>
<tr>
<td>1988</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>1989</td>
<td>15.0</td>
<td>3</td>
<td>73</td>
</tr>
<tr>
<td>1990</td>
<td>13.8</td>
<td>7.5</td>
<td>76</td>
</tr>
<tr>
<td>1991</td>
<td>10.0</td>
<td>19.1</td>
<td>95</td>
</tr>
<tr>
<td>1992</td>
<td>7.5</td>
<td>6.9</td>
<td>64</td>
</tr>
<tr>
<td>1993</td>
<td>10.7</td>
<td>3.3</td>
<td>68</td>
</tr>
<tr>
<td>1994</td>
<td>10.0</td>
<td>3.6</td>
<td>64</td>
</tr>
</tbody>
</table>

Table 7.9 Annual mean of NH₃-N in mg/l and percentage reductions in maturation ponds during the period 1988-1994.
From Tables 7.8 and 7.9, most of the NH₃-N reduction/transformations occur in the facultative stage. Although maturation pond 2.1 has consistently improved in % reduction efficiency there has been little change in that of the last maturation pond 2.2, however in 1994 there was a net % increase of NH₃-N in that particular pond. The overall system efficiency has improved slightly since the reduction in flow and salinity of the raw sewage.

7.3.2 Sulphate (SO₄) and Hydrogen Sulphide (H₂S) in WSP 1988-1994

The mean concentrations of SO₄ increased as the electrical conductivity increased and showed decreases as the repairs were made to the system.

In view of the reduced pond performance in faecal coliform removal efficiency and the BOD removals, the influence of elevated hydrogen sulphide levels is seen. As increased sulphide production adversely affects algal photosynthesis by reducing oxygen, and increasing turbidity of pond liquor it is not surprising that pond performance is adversely affected by the presence of high concentrations of hydrogen sulphides.

The concentrations of sulphates and sulphides measured in the WSP for each year of the monitoring programme are summarised in Table 7.10.
Table 7.10 Summary of annual means of $\text{SO}_4$ and $\text{H}_2\text{S}$ concentrations (mg/l) in raw sewage, septage, facultative and maturation ponds during the period 1988-1994.

<table>
<thead>
<tr>
<th>Year</th>
<th>IN SEW</th>
<th>PD 1.1</th>
<th>PD 1.2</th>
<th>PD 2.1</th>
<th>PD 2.2</th>
<th>SEPTAGE</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>$\text{SO}_4$</td>
<td>$\text{H}_2\text{S}$</td>
<td>$\text{SO}_4$</td>
<td>$\text{H}_2\text{S}$</td>
<td>$\text{SO}_4$</td>
<td>$\text{H}_2\text{S}$</td>
</tr>
<tr>
<td>1988</td>
<td>n/d</td>
<td>n/d</td>
<td>47</td>
<td>n/d</td>
<td>n/d</td>
<td>n/d</td>
</tr>
<tr>
<td>1989</td>
<td>223.9</td>
<td>9.8</td>
<td>162.0</td>
<td>1.1</td>
<td>161.5</td>
<td>1.0</td>
</tr>
<tr>
<td>1990</td>
<td>547.4</td>
<td>5.4</td>
<td>429.8</td>
<td>13.7</td>
<td>432.2</td>
<td>11.1</td>
</tr>
<tr>
<td>1991</td>
<td>855.2</td>
<td>6.1</td>
<td>743.4</td>
<td>13.3</td>
<td>739.4</td>
<td>15.8</td>
</tr>
<tr>
<td>1992</td>
<td>1355.5</td>
<td>4.7</td>
<td>1261.4</td>
<td>12.2</td>
<td>1237.2</td>
<td>14.6</td>
</tr>
<tr>
<td>1993</td>
<td>1145.5</td>
<td>5.3</td>
<td>1010.0</td>
<td>13.4</td>
<td>1054.0</td>
<td>12.6</td>
</tr>
<tr>
<td>1994</td>
<td>808.0</td>
<td>4.3</td>
<td>626.7</td>
<td>24.6</td>
<td>628.0</td>
<td>18.0</td>
</tr>
</tbody>
</table>
Table 7.11 Summary of annual mean of pH and temperature measurements taken during collection of samples for routine monitoring of WSP during the period 1988-1994.

<table>
<thead>
<tr>
<th>Year</th>
<th>IN SEW pH</th>
<th>IN SEW °C</th>
<th>PD 1.1 pH</th>
<th>PD 1.1 °C</th>
<th>PD 1.2 pH</th>
<th>PD 1.2 °C</th>
<th>PD 2.1 pH</th>
<th>PD 2.1 °C</th>
<th>PD 2.2 pH</th>
<th>PD 2.2 °C</th>
<th>SEPTAGE pH</th>
</tr>
</thead>
<tbody>
<tr>
<td>1988</td>
<td>7.4</td>
<td>29.2</td>
<td>8.59</td>
<td>28.5</td>
<td>8.77</td>
<td>28.3</td>
<td>9.02</td>
<td>29.1</td>
<td>9.17</td>
<td>27.8</td>
<td>n/d</td>
</tr>
<tr>
<td>1989</td>
<td>7.6</td>
<td>29.6</td>
<td>7.86</td>
<td>29.0</td>
<td>7.79</td>
<td>28.9</td>
<td>7.82</td>
<td>28.8</td>
<td>7.82</td>
<td>28.7</td>
<td>n/d</td>
</tr>
<tr>
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<td>7.5</td>
<td>30.2</td>
<td>7.80</td>
<td>29.2</td>
<td>7.74</td>
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<td>7.88</td>
<td>28.8</td>
<td>7.87</td>
<td>28.9</td>
<td>n/d</td>
</tr>
<tr>
<td>1991</td>
<td>7.6</td>
<td>29.4</td>
<td>8.00</td>
<td>28.5</td>
<td>7.96</td>
<td>28.6</td>
<td>8.08</td>
<td>28.4</td>
<td>8.09</td>
<td>28.5</td>
<td>7.41</td>
</tr>
<tr>
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<td>7.5</td>
<td>29.0</td>
<td>7.82</td>
<td>28.4</td>
<td>7.77</td>
<td>28.4</td>
<td>7.89</td>
<td>28.2</td>
<td>7.91</td>
<td>28.2</td>
<td>7.43</td>
</tr>
<tr>
<td>1993</td>
<td>7.6</td>
<td>29.5</td>
<td>7.68</td>
<td>28.6</td>
<td>7.70</td>
<td>28.7</td>
<td>7.81</td>
<td>28.6</td>
<td>7.89</td>
<td>28.5</td>
<td>7.30</td>
</tr>
<tr>
<td>1994</td>
<td>7.8</td>
<td>29.7</td>
<td>7.96</td>
<td>29.2</td>
<td>8.04</td>
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<td>8.13</td>
<td>29.2</td>
<td>8.16</td>
<td>28.9</td>
<td>6.69</td>
</tr>
</tbody>
</table>
7.3.3 Mean Temperature and pH Measured in WSP 1988-1994

In Chapter 6, temperature variations and stratification profiles were described and discussed. The data presented in Table 7.11 are averages of measurements taken at the time of routine sampling which was generally between 0900-1000 hours. The pH variations normally expected in WSP are not clearly shown in these data however there was a general increase throughout the system with the higher levels in the maturation ponds.

7.3.4 Suspended, Volatile and Fixed Solids Variations

Suspended solids (SS) throughout the WSP in Cayman is presented in Table 7.12. It is clear the these ponds are not effective at removing or reducing suspended matter. It was only in the first year of operation that substantial reductions were observed, and in the last year (1994) there was an average increase in SS by the time the final effluent left the pond.

Table 7.12 Annual average SS (mg/l) concentrations and percentage reductions in facultative ponds (1988-1994).

<table>
<thead>
<tr>
<th>Year</th>
<th>IN SEW</th>
<th>PD 1.1</th>
<th>% reduction</th>
<th>PD 1.2</th>
<th>% reduction</th>
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</thead>
<tbody>
<tr>
<td>1988</td>
<td>159</td>
<td>101</td>
<td>36</td>
<td>81</td>
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</tr>
<tr>
<td>1989</td>
<td>187</td>
<td>156</td>
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<td>1990</td>
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<td>1991</td>
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</tr>
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<td>1993</td>
<td>130</td>
<td>151</td>
<td>-16</td>
<td>168</td>
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<tr>
<td>1994</td>
<td>115</td>
<td>124</td>
<td>-8</td>
<td>137</td>
<td>-19</td>
</tr>
</tbody>
</table>
Table 7.13 Annual mean of SS (mg/l) and percentage reductions in maturation ponds during the period 1988-1994.

<table>
<thead>
<tr>
<th>Year</th>
<th>PD 2.1</th>
<th>% reductions</th>
<th>PD 2.2</th>
<th>% reductions</th>
<th>% reductions overall system</th>
</tr>
</thead>
<tbody>
<tr>
<td>1988</td>
<td>61</td>
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<td>49</td>
<td>20</td>
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<tr>
<td>1989</td>
<td>136</td>
<td>9</td>
<td>101</td>
<td>26</td>
<td>46</td>
</tr>
<tr>
<td>1990</td>
<td>167</td>
<td>2</td>
<td>160</td>
<td>4</td>
<td>12</td>
</tr>
<tr>
<td>1991</td>
<td>132</td>
<td>-15</td>
<td>133</td>
<td>-1</td>
<td>23</td>
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<tr>
<td>1992</td>
<td>119</td>
<td>-15</td>
<td>112</td>
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<td>35</td>
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<tr>
<td>1993</td>
<td>152</td>
<td>4</td>
<td>128</td>
<td>16</td>
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</tr>
<tr>
<td>1994</td>
<td>139</td>
<td>-7</td>
<td>138</td>
<td>1</td>
<td>-20</td>
</tr>
</tbody>
</table>

Monthly means of SS for all samples, except the septage sample, are plotted in Fig 7.4. From the results it is observed that the final effluent from maturation pond 2.2 is unable to meet the WA Regulations of 30 mg/l SS in effluents discharged to the environment. This of course should be seen in the context of the treatment method which is unlikely, even if performing properly, to meet the 30 mg/l effluent requirement.
Fig 7.4 Average monthly suspended solids (SS) of incoming sewage and all pond effluents. The final effluent is compared with WA Discharge Regulations.
7.4 Sludge Accumulation

The overall treatment processes taking place in WSPs include the anaerobic digestion of sedimneted biosolids and the deposition of undegradable solids, that is the accumulation of sludge.

The accumulation of sludge in WSP, as previously mentioned, affects nominal retention time by reducing the effective volume of the ponds as it builds up. Pearson et al (1987a) recommend that 5 points within each WSP be selected and the sludge measured (white towel method) in order to estimate the depth of sludge. This method was used in the Cayman ponds since Apr90, however in 1993 it was decided to increase the number of measurement points in each pond so that a more precise picture of the sludge accumulation pattern on the bottom of the pond.

In order to achieve this each pond was divided into 10m² grids (approx) and using a rope as a guide and to keep the boat stationary, 96 points were measured in each pond.

Data collected in Apr94 are plotted on Figs 7.5-7.12 to show the accumulation contours and topography of sludge accumulation. These isometric graphs demonstrating sludge accumulation are produced using the programme Surfer Version 6.

The detailed sampling strategy which was adopted to produce the views gives a fairly complete panorama of sludge topography and highlights the uneven nature of its distribution.

Fig 7.5 (facultative pond 1.1) clearly demonstrates the influence of the prevailing easterly wind on the deposition of solids on the westernmost wall. The plan view in Fig 7.6 reinforces the previous image and together they complement the hydraulic images presented in Chapter 10.

The value of this type of representation is that it makes it easier to visualise where, why and how short-circuiting can take place. Whereas it is obvious that a high proportion of solids are likely to be deposited near to the inlet, it could not necessarily have been predicted that a substantial quantity would also have been driven along the western wall towards the outlet.

This may also be indicative of the tracking route that is followed during short-circuiting i.e., close to the western wall. In pond 1.1 (Fig 7.5) this is likely to be aggravated by
the fact that 94% of the inlet flow is passing into the pond through the western of the two inlets as revealed by the study described and discussed in Chapter 10.

A vector analysis combining inlet flow direction and prevailing wind direction may confirm the resultant thrust along the western edge. A comparison of facultative pond 1.1, Fig 7.5 and pond 1.2, Fig 7.7 and 7.8 demonstrates a similar distribution of sludge with the exception of the fundamental difference conferred by the different quantities coming in through each of the 2 inlets, thus inlet solids deposition and hence flows seem to be equal in 1.2 and very unequal in 1.1.

Similarly useful observations can be made on the maturation ponds 2.1 and 2.2 (Figs 7.9-7.12). Thus in pond 2.1 sludge builds up again on the western side, but a majority of this amount is deposited on the downstream side of the baffle and an equally important fraction at the outlet end in two mounds. Similarly in pond 2.2, solids are deposited on the western side but most is deposited on the upstream side of the baffle. This deposit forms a ridge across the pond except in one deep saddle where short-circuiting almost certainly occurs.

The sludge of a facultative pond, if it is mechanically aerated, may take several years to reach steady-state conditions. However in the case on the CI ponds where artificial aeration was carried out for short periods it appears that the sludge accumulation has stabilised; this is indicated in the results shown in Table 7.14. The thickness of the sludge has in fact reduced since 1993. Samples of the sludge from each pond were collected in Mar94 and analysed for total % residue, % volatile and % fixed residue. The results are presented in Table 7.15.

Poduska *et al* (1981) reported that the sludge in WSPs exhibits redox conditions -200 to 300 mV which are conducive to sulphate reductions. Because of the excessive microbiological sulphate reduction that is known to take place in the CI system it is likely that the digestion of sludge at the bottom of the lagoons does not occur at the optimum or expected rate (high temperatures). This assumption may be made based on the knowledge that methane bacteria do not tolerate high concentrations of hydrogen sulphide as they are instead inhibited. The microbiological processes occurring under anaerobic conditions in sludge are presented in Fig 7.13.
Table 7.14 Mean sludge depth (m) in sewage treatment ponds during the period 1990-1994.

<table>
<thead>
<tr>
<th></th>
<th>PD 1.1</th>
<th>PD 1.2</th>
<th>PD 2.1</th>
<th>PD 2.2</th>
</tr>
</thead>
<tbody>
<tr>
<td>1990</td>
<td>0.145</td>
<td>0.164</td>
<td>0.054</td>
<td>0.041</td>
</tr>
<tr>
<td>1991</td>
<td>0.346</td>
<td>0.294</td>
<td>0.215</td>
<td>0.241</td>
</tr>
<tr>
<td>1992</td>
<td>0.385</td>
<td>0.362</td>
<td>0.177</td>
<td>0.217</td>
</tr>
<tr>
<td>1993</td>
<td>0.345</td>
<td>0.371</td>
<td>0.303</td>
<td>0.298</td>
</tr>
<tr>
<td>1994</td>
<td>0.343</td>
<td>0.345</td>
<td>0.123</td>
<td>0.126</td>
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</tbody>
</table>
Table 7.15 Analysis of sludge in WSP Mar94

<table>
<thead>
<tr>
<th>FOND</th>
<th>% TOTAL RESIDUE</th>
<th>% VOLATILE RESIDUE</th>
<th>% FIXED RESIDUE</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.1</td>
<td>4.77</td>
<td>58.81</td>
<td>41.19</td>
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<td>41.18</td>
</tr>
<tr>
<td>2.1</td>
<td>3.00</td>
<td>63.53</td>
<td>36.47</td>
</tr>
<tr>
<td>2.2</td>
<td>2.96</td>
<td>66.89</td>
<td>33.11</td>
</tr>
</tbody>
</table>
Fig 7.5 Topography of sludge accumulation in facultative pond 1.1 Apr94.
Fig 7.6 Sludge accumulation pattern/contour in facultative pond 1.1 Apr94.
Fig 7.7 Topography of sludge accumulation in facultative pond 1.2 Apr-94.
Fig 7.8 Sludge accumulation pattern/contour in facultative pond 1.2 Apr94.
Fig 7.9 Topography of sludge accumulation in maturation pond 2.1 Apr94.
Fig 7.10 Sludge accumulation pattern/contour in maturation pond 2.1 Apr94.
Fig 7.12 Sludge accumulation pattern/contour in maturation pond 2.2 Apr94.
7.5 Summary
By the end of this period, Dec94, the overall system removal of faecal coliform bacteria was still only 99.9% and the facultative ponds were removing only 98%. Thus, >10^4 cfu/100 ml faecal coliforms were frequently present in the final effluent. As the hydraulic overloading had been reduced considerably, the question remain, why does the system perform poorly in terms of faecal indicator bacteria removal? One of the most important factors that affects the removal of bacteria is retention time. In Chapter 10, a tracer study which was carried out to address these questions is described. Other influences such as salinity and sulphide levels are considered in Chapters 8 and 9.
CHAPTER 8

8.0 SALINE SEWAGE AND ITS IMPLICATIONS FOR TREATMENT AND REUSE

8.1 Saline Sewage - Discussion

From the monitoring programme and results presented in the 3 preceding chapters, the sewage treated in the CI sewage treatment lagoons may be characterised as urban domestic wastewater of low organic strength but with high salinity and sulphate content.

Although salinity has been shown in conventional treatment methods to negatively influence performance, this influence has been claimed to be low or negligible (Lawton and Eggert, 1957; Hall and Smallwood, 1967; Tokuz and Eckenfelder, 1978; and Poon et al.). However it has been shown through this monitoring programme that EC levels of >10000 μS/cm will negatively impact faecal coliform removals with hydrogen sulphide carry over to the maturation ponds and that BOD removal is slightly affected at the >5000 μS/cm levels (see Figs 8.1 and 8.2). It is the combination of sewage organics and sulphate that permit high levels of hydrogen sulphide to form from the metabolism of sulphate-reducing heterotrophs.

![Fig 8.1](image1.png)  
**Fig 8.1** Correlation between mean electrical conductivity and mean faecal coliform bacteria removal throughout the CI WSP, 1988-1994.

![Fig 8.2](image2.png)  
**Fig 8.2** Correlation between mean electrical conductivity and mean unfiltered BOD removal throughout the CI WSP, 1988-1994.
It is important to evaluate the effectiveness of biological wastewater treatment on saline sewage because as the world's population increases and there is the simultaneous increase in the demand for freshwater supplies, the use of seawater or brackish water as the carrier or flush water of domestic wastes may be a serious alternative some communities may have to consider. In particular, coastal communities, freshwater deficient or depleted tropical islands, and touristic communities may have little choice in the future if freshwater resources are poorly managed or are unavoidably depleted. In fact this situation has arisen in Charlotte Amalie, capital of the U.S. Virgin Islands, where seawater from the harbour is drawn and used for domestic flush water (Kessick and Manchen, 1976). Additionally, Dillaha et al (1986) suggested that as the islands of Micronesia suffer from severe freshwater shortages the only feasible carriage medium for waste is seawater or brackish groundwater.

The issue of saline wastewater treatment has been studied in some depth for rotating biological contactors (RBC's). RBC's have been reported in pilot studies to provide effective treatment of full-strength seawater to the same degree as non-saline sewage, producing an effluent suitable for discharge either to surface waters or in sea outfalls (Bishop and Kinner, 1981; and Kinner and Bishop, 1982). Poon et al (1979) also carried out a pilot-scale study on saline wastewater treatment with RBC's and concluded that the biological film on the rotating disks became acclimated to increased salinity, although there was a slight reduction in treatment efficiency.

In an attempt to resolve the conflict of evidence in the literature on the effects of high concentrations of inorganic solutes on the performance of activated sludge wastewater treatment systems, Tokuz and Eckenfelder (1978) performed experiments with saline wastewaters containing up to 50000 mg/l chloride concentrations. The results showed that activated sludges acclimatised to the high salinities and high organic loadings, although the BOD removal in the effluent declined by only 5%.

The effect of various salinity levels have been studied in completely mixed aerated systems and trickling filters by a number of researchers (Lawton and Eggert, 1957; Ludzak and Noran, 1956; and Kincannon and Gaudy, 1968). Generally the studies indicate that sewage treatment using biological treatment plants is feasible because the bacteria involved are capable of acclimatising to elevated salinities. Treatment proceeded at lower rates than in freshwater, however the removal rates were considered satisfactory. The major deterioration in treatment efficiency occurred when the system experienced abrupt changes in salinity, gradual increases did not seriously limit treatment.
From these studies it would appear that treatment of saline wastewaters by conventional treatment and stabilisation pond would be suitable processes to produce acceptable effluents for river or sea outfall discharge. It should be noted that none of the previously mentioned studies reviewed indicated faecal coliform % removals or odour problems. Possibly bacterial removals were not considered since the effluent obviously could not be reused for irrigation and would be discharged to receiving waters. The microbiological processes influenced by seawater will be further discussed and reviewed in this chapter.

8.2 Microbiology and the Sulphur Cycle

In order to rationalise the sulphate reduction occurring in the facultative lagoons, an understanding of the basic microbiology and chemistry involved in the biological sulphur cycle is fundamental. This cycle is described by Postgate (1984):

"Sulphate (SO$_4^{2-}$) is reduced to sulphide (S$^2$) by dissimilatory sulphate-reducing bacteria and provides substrates for sulphide-oxidising bacteria which convert it by way of elemental sulphur (S$^0$) back to sulphate. In assimilatory sulphate reduction, the sulphur of sulphate passes through the sulphide level of oxidation and becomes incorporated into an amino acid (RSH) before being built into plant as microbial protein. This is eaten by animals and the sulphur is eventually returned to the cycle as sulphide formed during the breakdown and putrefaction (by bacteria) of dead organisms"

Fig 8.3 is a diagrammatic representation of the biological sulphur cycle (Widdel, 1988):
Sulphate-reducing bacteria are found in natural waters of all salinities from zero to saturation (Postgate, 1984; Widdel, 1988; and Madigan, 1988). Although long lived and obligate anaerobes, sulphate-reducers are able to survive long periods in the presence of oxygen, but when anaerobic conditions return, resume sulphate-reducing activity. While anaerobic degradation in sulphate-poor environments such as freshwater habitats is dominated by methanogenesis, sulphate reduction dominates in marine or other sulphate-rich saline habitats (Widdel, 1988). Methanogenesis and sulphate reduction are regarded as alternate degradation reactions that compete for common substrates, favouring sulphate reduction if sulphate is present (Widdel, 1988).

In the presence of oxygen, denitrification, sulphate reduction and methanogenesis are suppressed. After oxygen is removed the reduction reactions occur in the order: denitrification, sulphate reduction and finally, methanogenesis.

Fig 8.3 illustrates that the reduction of sulphate in the facultative ponds is certainly as a result of bacterial conversion to sulphide. Metabolic activities of obligate anaerobes such as the Desulfovibrio, Desulfo bacter and Desulfonema species result in the generation of hydrogen sulphide in anaerobic environments.

Although sulphate-reducing bacteria have an almost universal distribution, their active ecology is generally limited to anaerobic environments. These microorganisms require
a low redox potential in order to multiply, thereby confining their activities to reducing environments. The most common representative in estuarine, brackish, and marine environments are strains of *Desulfovibrio* (Postgate, 1984). Conditions which favour *Desulfovibrio* are newly polluted or regularly polluted environments.

Waste stabilisation ponds provide the ideal niche for these micro-organisms to become established. Furthermore, the sewage influent provides a continuous inoculum of sulphate-reducing bacteria and organic material enabling the organisms to persist as a viable, active population. Once there is sufficient sulphate available and anaerobic conditions prevail in the pond liquid, they will flourish. Sources of *Desulfovibrio* and *Desulfotomaculum* include ruminant mammals and insect guts (Widdel and Pfennig, 1977) and also fresh human faeces (Postgate, 1984). The sewage lagoons in CI are characterised by having low DO, which, with the high sulphates in the raw sewage, provide an ideal environment for the stimulation of microbial sulphate reduction.

The major effects of massive biological sulphate reduction on the environment of the waste stabilisation pond eco-system may be summarised as:

1. Production of hydrogen sulphide which is toxic to indigenous microflora/fauna
2. Anaerobiosis
3. pH changes
4. Removal of organic matter
5. Alteration of microflora

### 8.2.1 Hydrogen Sulphide Generation and Toxicity

The overall metabolism of the sulphate-reducers may be described by two half-reactions of an oxidation-reduction pair. Sulphide is generated from the reduction of sulphate which serves as the final electron acceptor for the oxidation of organic substrate. During this process 8 electrons are transferred to sulphur as shown in the reduction half-reaction as follows (Poduska and Anderson, 1981):

\[
\text{SO}_4^{2-} + 8\text{H}^+ + 8e^- \rightarrow \text{S}^2^- + 4\text{H}_2\text{O} \quad \text{Eq 8.1}
\]

The metabolic energy required for cellular synthesis is conditional on the oxidation of the various organic substrates. The equation representing the overall reaction for oxidation by *Desulfovibrio desulfuricans* of methanol, for example, is:
The process by which the appropriate conditions for sulphate reducing bacteria are generated requires the initial activity of aerobic and facultative bacteria. These organisms will consume the available oxygen and in so doing create a more favourable, that is, reducing environment in which sulphate reducers can begin to multiply. These anoxic conditions will occur almost immediately in a turbid, organically rich wastewater if it is not actively aerated.

### 8.2.2 Anaerobiosis

The production of hydrogen sulphide has the most dramatic and damaging effect on the process of waste stabilisation in pond systems. Rapid activity of sulphate-reducers is stimulated by redox potential between -200 and -300mv. Then, as a result of hydrogen sulphide being generated the redox is reduced to -320mV, at which point free oxygen is undetectable except transiently. The optimal conditions for sulphide generation include a pH of 7-8 and temperature in the range 30-37°C (Baumgartner, 1934). Both these conditions are met in the primary ponds under study in Grand Cayman.

Due to the unavailability of oxygen, the activities of aerobic organisms will cease whilst those of anaerobes will be activated, thereby replacing the aerobic microflora with anaerobes. This results in organic matter being fermented rather than oxidised. In addition to suppressing the reproduction of aerobic microorganisms, sulphide is toxic to many of them. Higher macro-organisms are also susceptible to sulphide toxicity. Accordingly, most higher animals and plants will die, with their bodies increasing the amount of organic material available to the sulphate-reducing bacteria.

Hydrogen sulphide is known to be toxic to green algal species and potential predators, consequently, one of the major influences on 'stabilisation' of wastewater is disrupted in its presence. Hydrogen sulphide production in ponds evidently influences photosynthesis and consequently the removal of organic substrates. One of the few reported benefits of hydrogen sulphide is that at levels of 12 mg/l in anaerobic ponds studies have shown that *Vibrio cholerae* is reduced significantly (up to 75%) Pearson *et al* (in press a).
It was observed with the CI pond system that hydrogen sulphide levels of >1.0 mg/l were associated with brownish or pink coloured pond liquor, which in turn symbolised the absence of green algal species.

The terms 'green' and 'purple' bacteria are commonly used to describe the phototrophic bacteria known as the sulphide-converters. The phototrophs are characterised as having the ability to convert light energy to a chemically usable form through photometabolism in a strictly anoxygenic process (that is - molecular oxygen is not released as a photosynthetic by-product). The role of these organisms in aquatic habitats, is to reoxidise sulphides generating nontoxic species of sulphur (generally elemental sulphur), consequently creating a more favourable environment for the development of other organisms. Sewage treatment lagoons, especially facultative ponds, are environments where normally there are diurnal fluctuations between aerobic and anaerobic conditions. If there is a high concentration of sulphide, the growth of the phototrophs is enhanced.

The purple and green sulphur bacteria have distinct colouration which imparts to their aquatic environment a purple, red, pink, brown, yellow or even green colour. When sulphide-rich anaerobic systems are exposed to light, large blooms of the phototrophs may develop causing bright pigmentation in the system. Here again, sewage treatment ponds offer the ideal environment for the proliferation of the purple sulphur bacteria because of the presence of sulphate reducers and their by-product, hydrogen sulphide. Some studies have identified purple sulphur bacteria such as *Chromatium*, *Thiocapsa* and *Thiopedia* as important members of sewage ponds' microflora (Holm and Vennes, 1970; and Cooper *et al*, 1975).

In some of the earlier reports on the phenomenon of 'pink ponds' the colour was attributed to the 'pink algae', *Merisopedia* (Raman *et al*, 1970). Soler *et al* (*in press*) reported that in the Lorqui-Ceuti ponds, Spain, which have high sulphates due to fruit processing and cannery wastes mixed with the domestic sewage, that the purple-sulphur photosynthetic bacteria, *Thiopedia rosea* was the most common type responsible for the red colour of those ponds. This is commonly found in overloaded ponds (Gloyna, 1971; and Mara and Pearson, 1986). Evidently, by converting the hydrogen sulphide produced by sulphate-reducers, they can prevent or at least assist in odour control.

In Kenya, sewage treatment lagoons receiving domestic and industrial wastewater containing hydrogen sulphide (up to 40 mg/l, and theoretical retention >38.6 days) experienced poor faecal coliform bacteria removals, bright pink colouration even in the

Other studies (*Veenstra *et al.*, *in press*) on pond systems treating sewage strong in organic loading but low in sulphates attribute the typical reddish-pink colour in facultative ponds to the development of phototrophic purple non-sulphur bacteria, belonging to the genus of *Rhodopseudomonas*. A similar report was made by Jones (1956) in Minnesota, USA, where a bloom of *Rhodospirillaceae* was observed in highly loaded lagoon with a reduction in odour. This is not surprising as the normal habitat of the purple non-sulphur bacteria usually has little or no sulphate reduction. These microorganisms do not oxidise sulphide to sulphate but are associated with high turbidity in the upper water layers. This in turn, reduces the light penetration necessary for algal photosynthesis further creating conditions unfavourable to the growth of algae.

If the predators and green algal species that assist in creating a hostile environment for faecal indicator bacteria and pathogens are destroyed, then the efficiency of removing these organisms will as a result be lower. In the CI ponds the removal rates of the indicator faecal coliform bacteria have decreased in the whole system and in the maturation ponds as the electrical conductivity increased (Fig 8.2) and are further discussed in Chapter 11.

In Yemen, *Veenstra et al (in press)* reported that when high sulphides are present then a minimum retention time of 20-25 days in the facultative ponds is necessary to provide the ecological conditions that would allow algae to become re-established in the pond system.

Excessive hydrogen sulphide generation in sewage treatment ponds may result in serious odour problems. The most serious odour problems are usually reported in the early morning hours in the CI ponds which is in accordance with the findings of *Veenstra et al (in press)*. It also agrees with observations made in India (*Raman et al*, 1970) and more recently in the Alsamra ponds in Jordan (*Pescod, in press*) where reduction in hydrogen sulphide (offensive odours) during daylight hours was attributed to the oxidising of sulphides by phototrophic bacteria in the day (odour intensifies during the night when the phototrophs are inactive).
8.2.3 Changes in pH

During periods of active sulphate reduction, the process causes alkaline pH conditions to persist as the hydrogen sulphide is usually in the volatile state. However, simultaneous reactions occurring during methanogenesis lead to acid formation thereby counteracting the effect of sulphate-reduction. The net effect is near neutrality.

Chemical oxidation of sulphides occurs in the presence of oxygen but is pH dependent. Odour emissions are reduced if the pH is below 7.0 because undissociated H₂S dominates.

Fig 8.4 Effect of pH on hydrogen sulphide-sulphide equilibrium (32 mg H₂S/l).

At pH values above 7.0 HS⁻ is dominant (Fig 8.4) and is in the malodorous form. pH values above 9.5 cause reductions in odour emissions due to S²⁻ formation. Howsley and Pearson (1979) reported that sulphide toxicity to certain cyanobacteria is decreased with increased pH.
8.2.4 Removal of Organic Matter

A substantial quantity of organic matter is required for the process of sulphate-reduction, resulting in conversion to carbon dioxide. This allows a net mineralisation of organic matter to take place anaerobically.

8.2.5 Alteration of Microflora

As indicated earlier in this chapter, hydrogen sulphide is toxic to many organisms both micro and macro. Protozoa and rotifers do not survive very long in aquatic environments where anaerobic conditions coincide with the presence of high levels of hydrogen sulphide.

Hart et al (1990) reported that a freshwater aquatic system is able to withstand slow, small increases in salinity, but shock increases will have a deleterious effect. The consequence being that a lower (smaller) diversity of species in the biota will result as salinities increase. This is significant in the functioning of wastewater treatment lagoons, especially facultative and maturation ponds, because in them, the presence of micro-organisms such as protozoa and rotifers play an important part in the reduction of pathogenic organisms as they do in conventional treatment processes (Curds, 1973; Seaman et al, 1986; and Kawai et al, 1987). In Chapter 9, an investigation into the microbiota in the Cayman ponds is described.

8.3 Increased Salinity and Sulphate Concentrations

During the earlier stages of salinity problems, consideration was given to the possibility that septage being emptied into the system could have contributed to saline loadings. However, a 7 month (Jan91-Aug91) period of analyses of weekly composite septage samples did not reveal substantially excessive levels of salinity. As the total septage inflow compared to the incoming sewage flow contributes ≤1%, its influence is assumed to be negligible and, therefore was not considered further in terms of salinity problems.

It has been determined that the increased salinity in the raw sewage of the CI system is due mainly to infiltration into the sewers of salty groundwater (40000 μS/cm), which approaches the salinity of seawater (45000-55000 μS/cm) (Ng, 1989). The acceptance of seawater for toilet-flushing by 2 major hotels and the connection of the public beach facilities (which uses only seawater for lavatories and showers) resulted in additional increase of the raw sewage salinity.
As the electrical conductivity of the wastewater was measured prior to and after the connection of the saline dischargers, it was possible to estimate the percentage of salinity attributable to groundwater infiltration from the sewer system as presented in Table 8.1. The peak levels attained during 1992 are indicated in bold type. Previous data and observations reported in Chapters 3 and 5 identified saline groundwater intrusion as the cause for the failure in the integrity of the sewerage system.

<table>
<thead>
<tr>
<th>Year</th>
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<th>Saltwater % by Volume</th>
</tr>
</thead>
<tbody>
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<td>100</td>
</tr>
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<td>1992</td>
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<td>50</td>
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<tr>
<td>1990</td>
<td>12500</td>
<td>25</td>
</tr>
<tr>
<td>1989</td>
<td>6500</td>
<td>13</td>
</tr>
<tr>
<td>1988</td>
<td>5000</td>
<td>10</td>
</tr>
<tr>
<td>1988</td>
<td>2500</td>
<td>5 (normal)</td>
</tr>
</tbody>
</table>

It is estimated that in 1992 up to 45% of the raw sewage flow to the works was due to saline groundwater intrusion. This estimate was based on EC data available prior to the marked salinity increases experienced in Sep88. By assuming that Cayman sewage would normally have a salinity of approximately 2500 μS/cm, then sewage EC of 25000 μS/cm would suggest that approximately 45% of that salinity is attributable to infiltration of very saline groundwater. This assumption is born out in the correlations between increased flow and increased salinity. From data presented in Chapters 5 through 7 it is clear that the average daily flow increased simultaneously with the salinity. Economically, this was an undesirable situation as energy costs increased since there was more ‘sewage’ to pump. In addition as previously shown the treatment system’s performance efficiency was compromised and corrosion rates of concrete and steel structures rapidly increased. The rehabilitation of the sewerage system described briefly in Chapters 3 and 7, clearly was successful as it resulted in about a 40-45% reduction of flow (Fig 7.1, pg 153) and reduced pumping electrical costs (Fig 3.15, pg 60) by a similar percentage.

In order to compare the proportional contributions from constituents in saline waters it is necessary to know their typical principle concentrations present in a typical seawater
analysis. Table 8.2 presents typical cations and anions concentrations in seawater around the Cayman Islands.

<table>
<thead>
<tr>
<th>Constituent</th>
<th>Concentration in mg/l</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Cation</strong></td>
<td></td>
</tr>
<tr>
<td>Ca⁺⁺</td>
<td>1.200</td>
</tr>
<tr>
<td>Mg⁺⁺</td>
<td>5,350</td>
</tr>
<tr>
<td>Na⁺</td>
<td>23,540</td>
</tr>
<tr>
<td>K⁺</td>
<td>640</td>
</tr>
<tr>
<td><strong>Anion</strong></td>
<td></td>
</tr>
<tr>
<td>HCO₃⁻</td>
<td>132</td>
</tr>
<tr>
<td>Cl⁻</td>
<td>27485</td>
</tr>
<tr>
<td>SO₄²⁻</td>
<td>2800</td>
</tr>
<tr>
<td>pH</td>
<td>8.0-8.2</td>
</tr>
<tr>
<td>TOC</td>
<td>2</td>
</tr>
</tbody>
</table>

From Table 8.2, sulphate concentrations approximate 2800 mg/l for the seawater analysed from off the coast of Grand Cayman. Reported sulphate concentration in typical ocean water is 2700 mg/l (Turekian, 1969; and Riley and Chester, 1971). Laboratory analysis of the infiltrating saline groundwater has produced results in the range 1000-2000 mg/l (WAC laboratory data). Analysis of the incoming sewage revealed maximum sulphate concentrations of up to 1850 mg/l.

A graphical plot of electrical conductivity versus sulphate concentrations in the wastewater influent shows good correlation, $R^2 = 0.81$ (Fig 8.5).
This good correlation further supports the position that a rise in conductivity results in elevated sulphate concentrations if the cause of increased conductivity is attributable to saline waters. Several authors have reported similar experiences (Abbot, 1962; and Pinheiro et al, 1987) of poorly performing waste stabilisation ponds due to high salinities and sulphate-rich raw sewage.

The San Javier ponds in Spain (Soler et al, in press) also experienced infiltration of sea water into the water table. The pond system has had similar problems as the CI treatment ponds with a diluted sewage. High sulphate levels of 386-778 mg/l were reported by the authors, however they did not indicate odour problems in this system neither was there mention of ‘red or pink’ ponds as experienced in the CI system.

However, a bench-scale experiment with salinity levels ranging from 600-36000 mg/l (TDS) led Dillaha et al (1986) to conclude that waste stabilisation ponds were a viable option for treatment of saline wastewater in tropical areas. Nonetheless, they did acknowledge that lagoons without supplemental salt gave the highest performance in terms of BOD removal. At TDS levels of 600 mg/l, and loading of 12.5 kg BOD/ha day the % BOD removal was highest (97%). For the same loading and 36000 mg/l (TDS), the % BOD removal was 93% which is still an acceptable removal for facultative ponds. The % BOD removals for the CI system is much less ranging 80-83%. Oxygen was reported in these bench-scale studies to never drop below 4 mg/l.
The conclusions of Dillaha et al. (1986) may be somewhat premature as they were based on experiments where the influence of the anaerobic activity of sludge was not considered. It is inevitable that with the high sulphate concentrations and the optimum temperature present in the WSP that sulphide production would proceed at an accelerated rate, however, if the retention time is >20 days, a healthy algal population may develop.

### 8.4 Sulphur Conversions in CI Sewage Treatment Ponds

If aerobic conditions are maintained in biological wastewater treatment systems then the generation of offensive odours should be limited. As discussed earlier the presence of excessive amounts of sulphates in highly polluted aquatic environments such as wastewater treatment ponds provide ideal conditions for the development of anaerobiosis and the production of malodorous inorganic and organic compounds.

The odour problems identified at the CI sewage treatment works was clearly associated with the high concentrations of sulphate in the raw sewage. Emission of hydrogen sulphide to the atmosphere is governed mainly by the pH of the system in addition to the temperature, windspeed and direction, and mixing of the pond contents. As discussed in Chapter 3, mechanical aeration of the facultative ponds reduced the odour complaints from residents downwind, however, the trade-off was elevated energy costs.

The mean pH of the facultative ponds' effluents after the sulphate level increased to >300 mg/l was rarely >8.30. Although the presence of sulphide oxidising bacteria was obvious from the colour of the ponds, they were not sufficient to eliminate the emission of sulphide odours. This indicates that the conversion of sulphate to hydrogen sulphide proceeds at a higher rate than the oxidising of hydrogen sulphide by purple sulphur bacteria. It was suggested by Thistlethwayte (1972) that the rate of sulphide production is directly proportional to the sulphate concentration. Studies in the UK (Boon, 1992), however, did not reach the same conclusion finding instead that the rate of sulphide formation remained constant with varying sulphate concentrations.

The correlation coefficient between sulphate concentration and sulphide levels in the raw sewage is $R^2 = 0.8$. This fairly good correlation indicates that there is some dependence on the sulphate levels for the production of sulphide in sewage although the correlation between salinity (electrical conductivity) and sulphide production was not highly significant. The correlation coefficient between sulphate and sulphide levels in the effluent from facultative pond 1.1 was not highly significant, however in facultative pond 1.2 showed a correlation of $R^2 = 0.87$. 

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Average temperature of the facultative pond liquors is 29°C and ranges between 23-33°C, further enhancing conditions conducive to the growth of sulphate-reducers. The sulphate levels in the incoming sewage (grab samples) ranged from 173.3-1775.0 mg/l over the monitoring period 1989-1994.

The CI ponds are characterised by organic underloading and high sulphate concentrations above that recommended (<500 mg/l) in literature for WSP operation. Almasi and Pescod (in press) reported increased sulphide production in an anoxic pond as the sulphate increased from 100 mg/l to 325 mg/l, but further increase in sulphate to 550 mg/l had little effect on sulphide production. They reported that sulphide concentrations were highest when the BOD loading was high at 100 g/m³ day and was lowest when the BOD loading were low (30 g/m³ day). This was true at all sulphate concentrations.

Sulphide production in the Cayman WSP system cannot be attributed to organic overloading as the mean BOD volumetric loading is only 10 g/m³ day and ranges from 5-17 g/m³ day. Odour problems, nevertheless, were observed prior to the sulphate concentrations rising to the maximum recommended level, <500 mg/l. It is thus proposed for the Cayman Islands that the concentration of sulphate in raw sewage be no higher than 300 mg/l if the method of sewage treatment is WSPs.

Correlations between sulphates, sulphides and salinity were not significant in the effluents from the maturation ponds. The septage samples showed a wide variation in concentrations not highly correlated to salinity. Additionally, the influence of septage as previously mentioned was considered negligible due to its contribution in volume being <1% of the average daily flow.

8.5 Deterioration of Sewerage Structures
Concrete corrosion, one of the most serious problems facing sewerage systems, especially in the tropics, is due to environmental conditions that favour the development of the anaerobic and aerobic processes responsible for the formation of sulphuric acid (Heuer and Kaskens, 1993). This problem was further complicated in the CI system due to the excessive sulphates present in the sewage, causing concrete structures to be unavoidably prone to corrosion. Transformations of sulphur compounds is very significant when sulphate is excessive in sewage (Uhlmann, 1979a), affecting the entire sewerage system.
The entire CI sewerage system of pumping stations, manholes and electrical controls suffered severe deterioration due to the corrosive atmosphere occurring in the system. Maintenance problems associated with corrosive tropical marine environment gave rise to frequent and recurrent breakage of electrical components of the pumping stations’ electrical control panels. As the excessive sulphate was present due to the breakdown of sewerage system structures (Chapter 3), rehabilitation of the entire system was the solution chosen by the Authority.

8.5.1 Pumping Stations and Manholes - Concrete Corrosion

The breakdown in concrete structures in sewer systems is due to microbiological activity that ends in the production of sulphuric acid, a very aggressive acid. Sulphate reducing bacteria (such as *Desulfovibrio*) thrive only in the absence of oxygen. They do not utilise oxygen from the atmosphere but use the oxygen bound in sulphate, resulting in the production of hydrogen sulphide in the anaerobic, organic slime substrate that develops in sewer structures (Fig 8.6). Boon (1992) reported that only a small proportion of hydrogen sulphide produced in sewers occurs in the sewage, instead most of it is formed in this slime which grows on the walls. If the oxygen content in the sewage is low (<0.1 mg/l) and the pH is >7 but <9, then hydrogen sulphide is released to the atmosphere. Because of high humidity and warm temperatures water condensate forms on the walls and roofs of the sewer structures creating suitable conditions for the growth of autotrophic thiobacilli which oxidise hydrogen sulphide leading to the formation of sulphuric acid (Fig 8.6).

The optimum growth of *Thiobacillus neapolitanus*, usually found in the early stages of corrosion, occurs at pH 4-7. After the pH of original concrete surface (surface of fresh concrete is very alkaline at pH 12) is lowered to neutral due to carbonation, the *Thiobacillus neapolitanus* bacteria will become established. When these organisms, on account of sulphuric acid formation, have lowered surface pH to 4, the stage is set for the proliferation of the acidophilic bacteria, *Thiooxidans* and ferrooxidans increasing the pace of concrete deterioration. The sulphuric acid produced causes the cement mortar to breakdown. The main products resulting from corrosion is the formation of gypsum, at pH <3 and ettringite at pH >3.

The process occurring in the sewer pumping stations and manholes was dramatic and rapid. Although concrete specifications for the pumping stations and manholes required corrosion resistant Portland cement, less than 5 years after the system was commissioned these structures were deteriorating. This was not unusual given the damp, warm atmosphere in the system, and is similar to experiences reported in other parts of the world such as the Middle East, Australia, South Africa and the USA.

The structural degradation was such that up to 5 cm of a lead pencil could be inserted into the crumbling concrete walls of the interceptors. Therefore steps were taken by the WAC to rehabilitate the system as explained in Chapter 3. Mori et al (1992) reported corrosion rates of 4.3-4.7 mm/yr at the sewage water level of concrete structures (pipes); this meant that the life of the pipe would be about 20 years in Japanese conditions.

It is practically impossible to prevent the formation of hydrogen sulphide in the CI sewerage system because of the high sulphate concentration that is characteristic of the sewage. Nonetheless, minimisation of sulphide formation is desirable in order to reduce odour complaints and to protect the sewerage system, therefore all lift stations are fitted with air vents. The Authority has undertaken to replace the benching in manholes and pumping stations with corrosion resistant materials which also reduces the groundwater infiltration problem.
Fig 8.6 Schematic of corrosion processes in concrete sewer structures resulting from microbial sulphur transformations. \([H] = \text{organic substance as hydrogen donor.}\) (Modified and adapted from Uhlmann, 1979b).
8.5.2 Corrosion of Electrical Controls
Hydrogen sulphide will corrode metals such as copper, copper-based alloys such as brass. Consequently the gas can have a disastrous effect on electrical equipment at pumping stations (Boon, 1992). The controls in the CI system had to be replaced with specially ‘tropicalised’ controls less than 3 years after the system was commissioned because of the corrosive action of hydrogen sulphide gas escaping from the lift stations (Jagar, 1991).

8.6 Special Investigations
The substantial production of $\text{H}_2\text{S}$ is the most significant chemical outcome of the massive intrusion of salt water into the CI wastewater collection system. This factor led to the reformulation of the research objectives of this study and to the study of the functional ecology of the ponds described in Chapter 9 being curtailed. Early on in planning this study it had been proposed to study the mechanisms and ecological factors involved in pathogen and parasite removal and destruction. In Chapter 9 the preliminary work relating to this is described.
CHAPTER 9

9.0 STUDIES ON POND ECOLOGY, PARASITE REMOVAL & FAECAL INDICATOR DIE-OFF

9.1. Introduction
The primary focus of this section in the thesis is concerned with understanding the mechanisms affecting the performance of ponds in terms of pathogen removal. Three short-term studies were carried out.

First, an investigation into the functional ecology of WSP to establish a quantitative representation of certain groups of zooplankton and phytoplankton which could be used as complementary indicators of effluent quality was proposed. Specifically, to develop a biotic index similar to those used for surface water quality determination. The study commenced and after a number of microscopic examinations was discontinued due to treatment ponds' performance being complicated with the important practical problem of the continual rise in the salinity of the sewage. No baseline study could be undertaken to establish what a normal, healthy fauna should be. The author could only study the abnormal, high sulphide situation with depressed fauna. Secondly, a brief investigation into the loading of helminths to the ponds was carried out in order to assess the significance of this group in public health terms. Thirdly, the faecal coliform die-off rate for the CI ponds, under saline conditions was determined. This was used to identify appropriate $k_b$ values which may be useful for future local design purposes and for comparison with actual pond performance in terms of removal with design models.

The main objectives of the three studies reported and discussed in this chapter are:

<table>
<thead>
<tr>
<th>SPECIAL INVESTIGATIVE STUDY</th>
<th>MAIN OBJECTIVES</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Functional Ecology Assessment</td>
<td>a) Identification and quantification of protozoa and algae diversity</td>
</tr>
<tr>
<td></td>
<td>b) Estimation of contribution or effect on pathogen removal and/or predation</td>
</tr>
<tr>
<td></td>
<td>c) Evaluate feasibility of developing a biotic index</td>
</tr>
<tr>
<td>2. Ova/parasite</td>
<td>Preliminary survey for general evaluation of helminth loading to ponds</td>
</tr>
<tr>
<td>3. Pathogen Indicator Bacteria Die-off</td>
<td>Calculate faecal coliform bacteria die-off rate ($k_b$) for local conditions</td>
</tr>
</tbody>
</table>
9.2 Functional Ecology Assessment

This study was expected to assist in understanding wastewater lagoon treatment mechanisms and ecosystems dynamics, and to aid plant management through the development of a biotic index of effluent quality. It is important to note that pond BODs often deviate from calculated removal efficiency due mainly to stochastic short term variations in hydro-dynamic conditions, in addition to the fact that biological processes are almost permanently in a transient stage. As aerobic heterotrophic bacteria and microbiota become more diverse, the maturity of the lagoon as an autonomous community is demonstrated. The progress of the self-purification process in wastewater treatment ponds can be evidenced through the study of the functional biota. The functional ecology of WSPs involve the aquatic phyto and zooplankton found in ‘polluted’ waters.

It was intended to investigate the functional ecology and the influence of the relevant organisms (algae, protozoans, rotifers) on the behaviour of the ponds in terms of performance. The development of a biotic index using these organisms to predict pond effluent quality was seen as a tool that could be useful in pond systems within the region. In some conventional systems, much work has been done on the diversity and quantity of microflora at various stages and this data has been used to establish quality and predict performance for those biological wastewater treatment methods (Curds 1969 and 1973).

In fresh water systems (rivers, lakes) a biotic index is developed from consideration of 2 effects of pollution: reduction in community diversity; and the progressive loss of certain groups in response to organic pollution. Whilst in polluted aquatic ecosystems the reverse is true: as water becomes cleaner with less organic pollution, the diversity of the biotic community increases and there is a progressive appearance of certain groups.

A quantitative representation of certain groups of zooplankton and phytoplankton may be used as indicators as they are able to survive under certain conditions which are measurable. For instance most zooplankton cannot survive in the presence of hydrogen sulphide therefore if they are absent and certain types of phytoplankton are present then it may be possible to make assumptions regarding the quality of the aquatic environment.

If in addition to the biochemical and bacteriological monitoring parameters, examination of predators such as protozoa and rotifers is undertaken, enhanced understanding of lagoon behaviour in the context of bacteria removal mechanisms is possible. A survey of available literature revealed that there is not much known about the functional
ecology of wastewater lagoons in terms of pathogen removal by predators. It is known that in conventional aerobic treatment of sewage, protozoa play a vitally important role in the removal of dispersed bacteria (Curds, 1969). Predation by protozoa and rotifera is one removal method that has not received significant research attention, although it is assumed that predation accounts for a considerable reduction in bacteria (e.g. when a *Paramecium caudatum* filters particles 0.36 μm in size it will ingest about 400 particles per second) (Fenchel, 1980a). Additionally the ciliated protozoans play a significant role as consumers of phytoplankton (Fenchel, 1980b). There has been no 'in-depth' attempt to determine if protozoa as a whole influence faecal coliform removal in waste stabilisation ponds.

In order to relate results of micro-community and pond effluent quality, a standard sampling method must be devised. This would involve extensive study at different times of the day, different depths, whether composite column samples or grab samples were appropriate, identification of suitable sampling points in the ponds, and so on. Unfortunately due to the increasing salinity and the limited pond biota diversity it was no longer feasible to continue this study with the original objective in mind. Nevertheless the preliminary investigation was carried out and it is those results that are presented and discussed in the following section of this chapter.

### 9.2.1 Literature Review

There have been many academic studies on the variety of algal (phytoplankton) species (Singh and Saxena, 1969; Ahmed, 1974; Goulden, 1976; and Shillinglaw and Pieterse, 1977) to be found in WSP systems however none of those reviewed attempted to establish a biotic index using the micro-biota to predict effluent quality. A striking similarity between algal species present in facultative ponds throughout the USA led Porges and Mackenthun (1963) to conclude that geographical location had little effect on speciation. Palmer (1962) estimates that >20000 species of algae exist and reproduce in aqueous environments. However in WSP because of the pond liquid characteristics the number of common algal species is usually limited to around 25. He defined a number of groups as shown in Table 9.1:
Table 9.1 Predominant photosynthetic groups normally present in waste stabilisation ponds.
(Adapted from Palmer, 1962).

<table>
<thead>
<tr>
<th>Characteristics</th>
<th>Blue-green Bacteria</th>
<th>PHYTOPLANKTON GROUP</th>
<th>Pigmented Flagellates</th>
</tr>
</thead>
<tbody>
<tr>
<td>Colour</td>
<td>blue-green to brown</td>
<td>green to yellow-green</td>
<td>brown to light green</td>
</tr>
<tr>
<td>Location of photosynthetic pigment</td>
<td>throughout cells</td>
<td>in plastids</td>
<td>in plastids</td>
</tr>
<tr>
<td>Starch</td>
<td>absent</td>
<td>absent</td>
<td>absent</td>
</tr>
<tr>
<td>Cell wall</td>
<td>inseparable from slimy coating</td>
<td>semi-rigid or with spines</td>
<td>very rigid with regular marking</td>
</tr>
<tr>
<td>Nucleus</td>
<td>absent</td>
<td>present</td>
<td>present</td>
</tr>
<tr>
<td>Flagellum</td>
<td>absent</td>
<td>absent</td>
<td>absent</td>
</tr>
<tr>
<td>Eye spot</td>
<td>absent</td>
<td>absent</td>
<td>absent</td>
</tr>
</tbody>
</table>

Algae are an important feature and characteristic of healthy operating wastewater pond treatment systems. They contain chlorophyll and exhibit true photosynthesis utilising light as the energy source for cell synthesis. It is through the photosynthesis process that simple, stable inorganic compounds are converted into energy-rich organic matter (algal cells) and oxygen. This process is dependent on environmental conditions in the pond which must be conducive to the growth and development of healthy algal communities. The light intensity (solar radiation) and temperature are two of the major environmental factors that affect algae. The solar energy available for photosynthesis is a function of the geographical location, elevation, and meteorological conditions. Algae are able to survive in a wide temperature range: 4-40°C.

The biological activity in wastewater lagoons is greatly affected by the pH. From McGauhey (1968), photosynthesis oxygenation in ponds will occur more readily when the pH range is 6.5-10.5, all other factors being equal. Algae utilise CO₂ from the natural carbonate buffer system during peak photosynthetic activity. The resulting hydroxyl ions (OH⁻) cause the increase in pH found especially in maturation ponds during daylight hours. The milky appearance which occurs in some ponds is normally caused by the precipitation of CaCO₃ at the high pH values that may occur. Algal organisms exert influence on pond liquid by affecting pH alkalinity, hardness, turbidity and colour (Palmer, 1962).
The growth of any particular phytoplankton species is governed by a complex interaction of parameters such as temperature, light intensity, nutrients, salinity, pH mixing, and pre-adaptation (Goldman and Stanley, 1974; Richmond, 1983; and Oswald, 1988). In WSP systems the importance of the development of a healthy algal population has been well documented (Troussellier et al, 1986; and Troussellier and Legendre, 1989).

In order for healthy populations to develop, macro-nutrients such as carbon, hydrogen, oxygen, nitrogen, potassium and phosphorus must be present. Minute quantities of 'micro' - nutrients are also necessary; iron, magnesium, calcium, boron, zinc, copper, manganese, cobalt and molybdenum.

Two chemical compounds (ammonia and hydrogen sulphide) found in WSP or produced therein affect algae by inhibiting the establishment of significant populations. Both compounds are toxic to algae in the undissociated forms with the level of toxicity being pH dependent. The toxicity of sulphide increases as pH decreases while that of ammonia increases as pH increases. Clearly the efficiency of photosynthesis will be compromised by the presence of either chemical in the pond. Either sulphide or ammonia toxicity can result in rapid reduction in phyto- and zoo-plankton growth and the loss of of intolerant species, consequently oxygen production is reduced and faecal indicator and pathogen removal is likely to be compromised.

The toxicity of ammonia to algae, aquatic animals and plants in its unionised (NH₃) form (Fig 9.1) has been well described by Tabata, (1962), Warren, (1962) Natarajan, (1970), Abelovich and Azov (1976), Konig et al (1987), and Pearson et al (1987c).

Ammonia is toxic because being lipid soluble and uncharged it is able to cross biological membranes easier than charged and hydrated ammonium (NH₄) ions. Chlorella is one of the species that is very tolerant to high levels of ammonia even up to 600 mg N-[NH₃+NH₄⁺]/l at pH 8-9 as reported by Matusiak (1977). This was confirmed in a study carried out by Pearson et al (1987c) where it was reported that 50% of Chlorella photosynthesis was inhibited at pH 8.5 by 356 mg/l ammonia. Scenedesmus showed the same amount of photosynthetic reduction at 150 mg/l. The Euglena and Chlamydomonas sp were less tolerant exhibiting 50% reduced photosynthesis at 81 mg/l and 87 mg/l respectively. Abeliovich and Azov (1976) reported that Scenedesmus obliquus, Chlorella pyrenoidosa and Anacystis nidulans (cyanobacteria) showed 50% reduction in photosynthetic activity in a 0.675 mM free NH₃ concentration.
In Kenya, sewage treatment lagoons receiving domestic and industrial wastewater containing hydrogen sulphide (up to 40 mg/l, and theoretical retention >38.6 days) experienced poor faecal coliform bacteria removals, bright pink colouration even in the final maturation ponds, absence of oxygen, and the presence of the purple photosynthetic bacteria, *Thiopedia rosea* (Alabaster *et al*, 1991). *Chlorella sp* were the algal species most frequently observed in the ponds’ effluents although *Euglena, Phacus*, and *Micratinium* were also identified. *Chlorella* is the most resistant, of the 4 algae species identified, to sulphide toxicity, however since *Euglena* and *Phacus* are motile they are able to escape to the top water level of the pond and survive (Pearson *et al*, 1987c). Only *Chlamydomonas*, an alga known to be tolerant to polluting conditions, was found in the highest loaded facultative ponds (Alabaster *et al*, 1991). The Pearson *et al* (1987c) study, however reported that in WSP systems *Chlorella* is the alga most tolerant of sulphide toxicity.

A search of relevant literature was carried out specifically to review studies on saline sewage because the CI ponds are characterised by high salinity, however only a few were found. In continuous culture experiments of seawater mixed with secondary sewage effluent (in 15 l batches and 400 l pilot-scale ponds), Dunstan and Menzel
(1971) and Dunstan and Tenore (1972) reported that only 1-4 species of algae normally present in seawater remain after 2-3 weeks. Goldman et al (1974) reported the dominance of the *Bacillariophyceae sp* (marine algae) in a continuous 2000 l pond. Although these investigations were related to the effectiveness of algae in reducing nutrients (N and P) in sewage effluents discharged out to sea outfalls, the similarity with the CI system is valid because of the low organic strength of the sewage and the salinity levels.

The ‘blue green ‘algae’’ are in fact bacteria as their DNA is not delimited by a membrane, their photosynthetic pigment is not membrane bounded and their cell walls are those of bacteria. They possess all the charactereristic features of prokaryotes. However unlike other photosynthetic bacteria, the ‘blue greens’ are capable of synthesising oxygen. The group is of particular interest in the context of the present study because all blue greens have the ability to utilise ammonia and many are tolerant of hydrogen sulphide (H₂S).

Benson-Evans and Williams (1975) have summarised the indicator biotic significance of a number of species using the saprobic classification system. Thus *Oscillatoria lauterbornii* is classed as a polysaprobic and occurring in high H₂S environments; *O. limnosa* as associated with sulphur-bacteria communities in sewage; *O. putrida* as frequent in waters with H₂S and facultatively anaerobic, whilst other species of *Oscillatoria* tend to rise up into the plankton zone when H₂S increases in the lower water layer. The importance of this for the Cayman ponds is that like the purple sulphur bacteria, some blue greens, for example, *O. limnetica* can carry out anoxygenic photosynthesis using H₂S which is oxidised to sulphur, whereas in the dark the sulphur can be converted back to sulphide.

Dillaha et al (1986), in bench scale experiments investigating the feasibility of designing WSP systems to treat saline sewage, reported that the blue-green algae (*Anacystis* and *Nostoc* genera) dominated in saline experiments (12000-22000 mg/l TDS), although there were some green algae observed (*Chlorococcum* and *Palmella*). In the freshwater control experiment, a significant population of protozoans, crustaceans and insect larvae developed. Experiments with increasing salinity exhibited a decreasing variety of species in the microflora.

The population dynamics of the higher protists and small animals in pond systems has not been studied extensively because their role in pond treatment has been considered secondary to the algal/bacterial symbiotic interactions (Mitchell, 1980b). However grazing pressure by *Daphnia* is known to terminate algal blooms resulting in good
quality effluents (Mitchell and Williams, 1982). Curds (1982) pointed out that the majority of authors in publications on the role of protozoa in the purification of sewage agree that when large numbers of ciliates are present the effluent is 'clear and sparkling'.

Studies on zooplankton in ponds (Wennstrom, 1955; and Afanes et al, 1981) found protozoa to be dominant in the cold months whereas multicellular animals grew readily during the remainder of the year. Paramecium was reported as the primary free swimming ciliated protozoan while Glaucoma, Euplotes and Colpidium were other types also found in large numbers. Rotifers and the crustacean, Daphnia were found to be active in the summer months. Daphnia ate most of the algae in the final pond producing an effluent essentially free of algae. The rotifers will feed on bacteria and Cladocera.

9.2.2 Methods for Identification of Phyto and Zooplankton

Grab samples were collected from the CI ponds on 5 occasions for analysis. Two of the samples were collected in the dry season (Feb91 and Mar91) when the ambient air temperature is lower, and the remaining were collected weekly for 3 consecutive weeks in Jun91 (wet season). The samples were collected in dry-oven sterilised 1 1 glass bottles and transported to the laboratory within 40 mins.

Analysis for these organisms was by direct examination of approximately 1.0 μl of pond effluent grab samples with an improved Neubauer cell under a Bausch and Lomb microscope using magnification objective 40x/100x and a 10x eyepiece. An ocular micrometer was used to estimate cell sizes. It has been suggested that when using the direct examination method it is easier to identify protozoa although identification of some flagellate categories and small abundant protists may not be estimated reliably (Baldock, 1986). However as this was a preliminary survey, it was resolved to use the simplest and least time-consuming method. A minimum of 3 slides were prepared for each sample. The arithmetic mean of the triplicate results for each sample was calculated and reported as number of organisms/0.1ml.

No stains were used, neither were the organisms' movements inhibited by chemical manipulation. Identification was done using line drawings, keys and photoplates in Donner (1966), Curds (1969), Bick (1972), Uhlmann (1979b), Fox et al (1981) and APHA Standard Methods (APHA, 1985). Samples were examined no later than 3 hours after collection.
9.2.2.1 Chlorophyll a

Certain pond samples were analysed for chlorophyll a to determine productivity. The methanol extract method used is described by Pearson et al (1987a). The samples were filtered using 25 mm glass-fibre filter discs (Whatman GF/C), followed by extraction with 90% methanol, and centrifuging at 500-600 g. The absorbency of the supernatant was measured at 663 nm and at 750 nm using a 1 cm cuvette with a Bausch and Lomb mini-20 spectrophotometer. The absorbency readings were generally lower than the range recommended (0.2-0.8 abs) by Pearson et al (1987c), however the sample volumes filtered were the maximum capable of going through the filter paper. The mean chlorophyll a was determined from sample duplicates. All samples were analysed within 3 hours on the same day they were collected.

9.2.3 Results and Discussion

Normally, this type of investigation into the functional ecology of WSP would be carried out on a frequent and regular basis throughout the wet and dry seasons. However that was not possible in the course of this preliminary work and in the absence of more data, the results obtained are discussed in this section.

The raw sewage was examined on two occasions in Feb91 and Mar91. *Euglena* (266/0.1ml), *Chlorella* (33/0.1ml) and *Spathidium* (66/0.1ml) were found in the samples in Feb91. As the WSP system is operated with approximately 20% recirculation of the effluent from pond 2.2 to the inlet works it is expected to find these organisms in the raw sewage at times, however none were detected in the Mar91 samples.

During this survey the aerators in the facultative ponds were not operated. Both facultative ponds' liquid were characterised by a pinkish/brown milky appearance. The microscopic examination of all 'pinkish' samples revealed the presence of blue green bacteria often incorrectly referred to as 'algae'. Their morphological characteristics were indistinguishable from *Merismopedia* and were provisionally assigned to this genus. The residue remaining on the filter paper after filtration for the chlorophyll a analysis was more pinkish than green indicating that the chlorophyll concentration was not very high. This is seen in the results where the maximum chlorophyll measured was <1000 µg/l (see Table 9.5, maturation pond 2.2).

Considerable variation in the total algal cell numbers was found during the limited sampling period (Tables 9.2-9.5). Total cells in facultative pond 1.1 ranged from 1525 to 2830/0.1 ml; in facultative pond 1.2 total cells ranged from 400 to 3050/0.1 ml.
These levels were in the lower end of the range reported by (Irving and Dromgoole, 1986) of 19-725 x 10^3 cells/ml in facultative ponds in New Zealand; and by Rao (1983) who reported counts of 33.3-216.7 x 10^3/ml in primary ponds in India.

In the maturation pond 2.1 the total cells ranged from 540-2930/0.1 ml; the final maturation pond 2.2 had a range of 150-2357/0.1 ml for the 5 sampling periods. These concentrations are low in comparison to the counts reported in the Indian study (Rao, 1983) in a secondary pond. Because the number of samples points are low, discussions and conclusions are made with caution. The surface loading of BODuf, concentrations of, NH_3-N, H_2S and the pH and temperature for each pond during the time the samples were analysed are shown in Tables 9.2 to 9.5 with the results of the survey.

Table 9.2 Functional ecology survey results and selected physico-chemical parameters in facultative pond 1.1 effluent at the time of sampling.

<table>
<thead>
<tr>
<th>Date</th>
<th>Temp °C</th>
<th>pH units</th>
<th>BODuf kg/ha day</th>
<th>NH_3-N mg/l</th>
<th>SS mg/l</th>
<th>H_2S mg/l</th>
<th>Chlorophyll a µg/l</th>
<th>Phyto/zoo-plankton present cells /0.1 ml</th>
<th>Rotifers present /0.1 ml</th>
</tr>
</thead>
<tbody>
<tr>
<td>5 Feb91</td>
<td>26.0</td>
<td>8.20</td>
<td>149.6</td>
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<td>12.5</td>
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<td>nil</td>
</tr>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
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<td>8.33</td>
<td>206.9</td>
<td>not done</td>
<td>114.2</td>
<td>7.5</td>
<td>not done</td>
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<td></td>
<td></td>
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<td>Eutrochalea sp</td>
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<td></td>
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<td>Chlorella sp</td>
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<td>204.9</td>
<td>8.0</td>
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<td>7.5</td>
<td>260.0</td>
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<td>110.2</td>
<td>15.0</td>
<td>351.0</td>
<td>Eutrochalea sp</td>
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</tr>
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<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td>Merismopedia sp</td>
<td>2550</td>
</tr>
</tbody>
</table>

In the facultative pond 1.1 (Table 9.2), seven different species of algae were identified and reported. The most persistent organism found in all samples had the characteristic morphology of Merismopedia, however its pigmentation was uniformly pink and no reference was found to indicate whether what is classified as a blue green can have a red photosynthetic pigment. However sulphide allows for anoxygenic photosynthesis.
by cyanobacteria (Cohen et al., 1986) and such organisms may well have a 'red' photosynthetic pigment as well as the 'purple' sulphur bacteria.

Species diversity was markedly reduced between the samples taken on 5 Feb91 and those on 20 Jun91 (Fig. 9.3). The most notable difference in the chemical characteristics of the pond on the two dates was the increased salinity.

Fig 9.2 Diversity of and percentage zoo/phytoplankton species in facultative pond 1.1 effluent on 5 Feb91 and 20 Jun91 (H2S = 12.5 mg/l and 15.0 mg/l, respectively; and EC = 11630 μS/cm and 18400 μS/cm, respectively).

Data from facultative pond 1.2 is presented in Table 9.3:
Table 9.3 Functional ecology survey results and selected physico-chemical parameters in facultative pond 1.2 effluent at the time of sampling.

<table>
<thead>
<tr>
<th>FACULTATIVE POND 1.2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Date</td>
</tr>
<tr>
<td>------</td>
</tr>
<tr>
<td>5 Feb91</td>
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<tr>
<td></td>
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<tr>
<td></td>
</tr>
<tr>
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<tr>
<td>4 Jun91</td>
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<tr>
<td>12 Jun91</td>
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<tr>
<td>20 Jun91</td>
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<td></td>
</tr>
</tbody>
</table>

The influence of hydrogen sulphide inhibition is not as pronounced in the pond 1.2 data (Table 9.3), where even with concentrations as high as 50 mg/l Euglena and Chlorella sp persisted. There is no explanation for the higher sulphide levels found in pond 1.2 compared to pond 1.1 as both ponds received equal organic loading and operated under the same conditions. In the facultative pond 1.2, six species of zoo/phytoplankton were identified and reported. As in pond 1.1, the dominant species was the blue-green 'algae'; Merismopedia sp.

Data from maturation pond 2.1 is presented in Table 9.4:

Table 9.4 Functional ecology survey results and selected physico-chemical parameters in facultative pond 2.1 effluent at the time of sampling.

<table>
<thead>
<tr>
<th>MATURATION POND 2.1</th>
</tr>
</thead>
<tbody>
<tr>
<td>Date</td>
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<td>-------</td>
</tr>
<tr>
<td>5 Feb91</td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td>25 Mar91</td>
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</tr>
</tbody>
</table>

216
Even though the sulphide concentration in effluent from maturation pond 2.1 did not exceed 5 mg/l on sampling days, only six species were isolated and reported. However, the *Euglena sp* dominated for the first two samplings and in the last sample the dominant species was the blue-green algae; *Merismopedia sp*.

Data from maturation pond 2.2 is presented in Table 9.5:

<table>
<thead>
<tr>
<th>Date</th>
<th>Temp °C</th>
<th>pH units</th>
<th>BODuf kg/ha day</th>
<th>NH₄-N mg/l</th>
<th>SS mg/l</th>
<th>H₂S mg/l</th>
<th>Chlorophyll a µg/l</th>
<th>Phyto/zooplankton present cells/0.1 ml</th>
<th>Rotifer present/0.1 ml</th>
</tr>
</thead>
<tbody>
<tr>
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<td>26.0</td>
<td>8.04</td>
<td>87.8</td>
<td>8.8</td>
<td>142.4</td>
<td>0.0</td>
<td>not done</td>
<td><em>Euglena sp</em></td>
<td>1400</td>
</tr>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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</tr>
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<td></td>
<td></td>
<td></td>
<td></td>
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</tr>
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<td></td>
<td></td>
<td></td>
<td></td>
<td><em>Spathidium sp</em></td>
<td>300</td>
</tr>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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</tr>
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<td></td>
<td></td>
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<td></td>
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<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td><em>Merismopedia sp</em></td>
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</tr>
<tr>
<td>25 Mar91</td>
<td>29.0</td>
<td>8.04</td>
<td>103.8</td>
<td>8.0</td>
<td>164.1</td>
<td>0.0</td>
<td>not done</td>
<td><em>Euglena sp</em></td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td><em>Lionotus sp</em></td>
<td>25</td>
</tr>
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<td></td>
<td></td>
<td></td>
<td></td>
<td><em>Euplotes sp</em></td>
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<tr>
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<td><em>Merismopedia sp</em></td>
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<td>12 Jun91</td>
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<td>87.2</td>
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<td>4.0</td>
<td>337.7</td>
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<td></td>
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<td></td>
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<td></td>
<td></td>
<td></td>
<td><em>Merismopedia sp</em></td>
<td>510</td>
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</table>

A total of 7 zoo/phytoplankton and 1 species of rotifer were identified in the final maturation pond 2.2. The blue-green, *Merismopedia sp* did not exhibit dominance until the last sample on 20 Jun91. The 3 samples taken prior, were dominated by the *Euglena sp*. The species diversity was also markedly reduced between the samples taken on 5 Feb91 and those on 20 Jun91 (Fig. 9.4). The most notable difference in the chemical characteristics of the pond on the two dates was the increased salinity and the presence of hydrogen sulphide.
The results above show the diversity reduction where blue-green 'algae' went from representing 1% of the total cell count in Feb91 to almost 90% dominance on 20 Jun91. Parker (1979) in his comprehensive review of biological mechanisms in lagoons observed that increased salinity tends to encourage blue-green 'algae' such as Anacystis and Oscillatoria to develop. Abbot (1962) reported that ponds at Muizenberg, South Africa, became heavily contaminated with seawater and consequently high sulphates (800 mg/l) and chlorides (1000 mg/l) in the sewage. He observed that they developed a pink colour which was attributed to Thioplycoccus, a sulphur bacterium. The high levels of chloride and sulphate completely changed the 'algal' flora of the ponds: Chlorococcales were displaced entirely by Euglenaceae. Sulphates were reduced to hydrogen sulphide which caused heavy mortality of the algae and further deoxygenation in the primary pond. The problem was resolved after the broken sewer was repaired. The pond was isolated and heavily chlorinated which did nothing to revive the algal population, therefore they just drained the pond! This is not a solution to the Cayman ponds.

Limited diversity and lower numbers are demonstrated in the data from the Cayman ponds compared to other studies. Patil et al (1975) reported 16 algal species identified in various densities in Indian facultative pond with an operating temperature of 25-34 °C, DO up to 6 mg/l, BOD removal average 30-90%. Twelve species of protozoans (mainly ciliated) were also identified in that study, the most common being Euglena sp
(16000-104000 cells/ml) and the most common alga was the *Chlorella sp* (16000-88000 cells/ml). Another study in Pakistan reportedly identified 37 algal species in WSP system in Pakistan (Ahmed, 1974). In India (Rao, 1983) reported 33 species belonging to 28 genera in the primary pond, while 37 species comprising 32 genera were found in the secondary pond. In Yemen ponds, Veenstra *et al* (*in press*) reported the green alga, *Euglena* as the dominant species.

Soler *et al* (1991) reported that the predominance of the *Chlamydomonas* species coincided with the proliferation of the purple sulphur bacteria *Thiocapsa sp* and *Chromatium sp*. In Kenya, (Alabaster *et al*, 1991) recorded *Chlorella sp* as the algal species most frequently observed. The Pearson *et al* (1987c) study, reported that in WSP systems *Chlorella* is the alga most tolerant of sulphide toxicity. *Euglena* and *Phacus* are more sensitive, however they are motile therefore able to escape to the top water level of the pond and survive. In Kenya, only *Chlamydomonas*, an alga known to be tolerant to polluting conditions, was found in the highest loaded facultative ponds (>1000 kg/ha day) (Alabaster *et al*, 1991). The Cayman facultative ponds, in contrast, were loaded at 93-212 kg/ha day during this study (Tables 9.2 and 9.3).

Table 9.6 gives a summary of the 9 species of phyto and zooplankton detected in the Cayman WSP during this investigation and the maximum levels of selected physico-chemical parameters at which they occurred.

<table>
<thead>
<tr>
<th>Microflora and fauna found in CI wastewater stabilisation ponds and simultaneous maximum levels of selected physico-chemical parameters</th>
<th>pH units</th>
<th>H₂S mg/l</th>
<th>NH₃ mg/l</th>
<th>Salinity EC µS/cm</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Agmenellum sp</em></td>
<td>7.7</td>
<td>12.5</td>
<td>8.0</td>
<td>17070</td>
</tr>
<tr>
<td><em>Chlorella sp</em></td>
<td>8.3</td>
<td>0-50</td>
<td>32</td>
<td>18400</td>
</tr>
<tr>
<td><em>Epiphanes sp</em></td>
<td>8.1</td>
<td>0</td>
<td>8.8</td>
<td>16080</td>
</tr>
<tr>
<td><em>Euglena sp</em></td>
<td>8.3</td>
<td>0-50</td>
<td>32</td>
<td>18400</td>
</tr>
<tr>
<td><em>Euplotes sp</em></td>
<td>8.2</td>
<td>0-22.5</td>
<td>32</td>
<td>17200</td>
</tr>
<tr>
<td><em>Litonotus sp</em></td>
<td>8.2</td>
<td>0-5</td>
<td>12</td>
<td>17600</td>
</tr>
<tr>
<td><em>Merismopedia sp</em></td>
<td>8.3</td>
<td>0-50</td>
<td>32</td>
<td>18400</td>
</tr>
<tr>
<td><em>Spathidium sp</em></td>
<td>8.3</td>
<td>0-7.5</td>
<td>8.8</td>
<td>16780</td>
</tr>
<tr>
<td><em>Tetrahymena sp</em></td>
<td>8.2</td>
<td>0-22.5</td>
<td>12.5</td>
<td>17200</td>
</tr>
</tbody>
</table>

In the CI facultative ponds the predominant species of alga are the *Merismopedia* sp which appears tolerant to sulphide levels of up to 50 mg/l, as does the *Chlorella sp* and
*Euglena sp* (Fig 9.5). Blue green 'algae' such as *Merismopedia sp* survive because they are able to metabolise under anaerobic conditions and utilise $\text{H}_2\text{S}$. In the maturation ponds the next dominant species is the *Chlorella*.

The protozoans such as *Tetrahymena* and *Euplotes* were less tolerant and were only found in samples where the $\text{H}_2\text{S}$ concentration was just over 20 mg/l. However, *Litonotus, Spathidum* and *Agmenellum* were not found in samples with sulphide >15 mg/l. The only species of rotifer identified, *Epiphanes*, was detected in samples with zero hydrogen sulphide mg/l (Fig 9.5).

![Fig. 9.5 Maximum concentration of hydrogen sulphide and the presence of phyto- and zooplankton in the Cayman WSP effluents in the period Feb91-Jun91.](image)

Abeliovich (1986) reports that in ponds experiencing high sulphide levels, the *Chlorella sp* is advantageous for oxygen supply as it maintains photosynthetic activity up to 1 mM sulphide concentration. Furthermore, Fruend *et al* (1993) concluded in their study that *Chlorella* was the most important alga in Israeli oxidation ponds because:

- high number present throughout year
- highest potential for oxygen supply per unit biomass
In contrast, investigative studies by Irving and Dromgoole (1986) indicate that the *Euglena sp* is probably responsible for the majority of oxygen produced in facultative ponds (New Zealand). However, the tolerance of *Euglena* to high sulphide levels in the Cayman ponds appears to be more than that proposed by Pearson *et al* (1987c). This report indicated development of >1 mg/l sulphide in surface waters will significantly reduce algal growth and oxygen generation by *Euglena* and could lead to algal washout and totally anaerobic conditions.

The influence of high/increased sulphide generation contributes to the drastic decline in algae in the CI ponds as is apparent from the colour changes observed; from vibrant green to brown to pink, and the low chlorophyll concentrations. Brockett (1977) observed that when a loss in algal activity occurred in facultative ponds it was accompanied by a milky opaque appearance and the production of H₂S. Pinheiro *et al* (1987) reported a similar experience in high-rate algal ponds in Alchote, Portugal where the predominant algae species *Euglena* and *Chlorella*, and occasionally *Chlamydomonas* and *Oscillatoria* virtually disappeared from the pond effluents when the ponds changed from green to pink in colour. This was attributed to massive growth of *Chromatiaceae* which requires and is more tolerant to the large amounts of microbially produced sulphide. Pinheiro *et al* (1987) identified sulphide concentrations of >20 mg/l and sulphate >600 mg/l as the limits of tolerance. In the CI ponds, sulphate levels ranged from 450 to 700 mg/l during this investigation.

The findings in the CI ponds during this investigation do not compare favourably to other studies as generally neither *Chlorella* or *Euglena* were predominant. The pink colour of the ponds appeared to be due to *Merismopedia sp*. This is a Cyanophyta or 'blue-green algae' and was conspicuous under the microscope because of its characteristic cellular arrangement and abundance. The 'pink phenomenon' observed in a facultative pond in India was caused by *Merismopedia tenuissima*, an 'alga' that has metabolism similar to that of sulphur bacteria (Raman *et al*, 1970). The phenomenon was favoured by reduced sunlight, high organic loads, increased depths, high temperature, and excessive sludge deposits. Blue green 'algae' are more adaptable to temperature extremes and chemical changes than the Chlorophytes (Benson-Evans and Williams, 1975). In the New Zealand facultative pond study, *Chorella sp* made up 80% of the total cell numbers and *Euglena sp* <10% of the total (Irving and Dromgoole, 1986) whereas in the Cayman lagoons *Chlorella sp* constituted between 1-62% and the
Euglena sp between 2-59%. Euglena sp and Chlorella sp were the predominant groups of algae found in facultative and maturation WSP in Sesimbra, Portugal (Mendes et al, in press).

Irving and Dromgoole (1986) established that the net oxygen production of Chlorella sp (found in brown ponds) was considerably less than that of Euglena. The Cayman ponds showed that Euglena sp prevailed over Chlorella sp.

The facultative ponds' effluent samples are representative of pond liquid at a depth of approximately 1.3 m while the maturation ponds' represent liquid at about 1.0 m depth. Chlorella sp are not as motile as the Euglenas which are known to migrate up and down the water in the ponds, avoiding intense light and toxic sulphides. The movement of Euglena sp is a plausible explanation for the lower numbers of Chlorella sp counted. This also demonstrates that in reporting pond biota data, it is essential to include information on sample depth, time and other environmental factors as these affect the diversity and numbers recovered.

Most bacteria feeding ciliate protozoa are quoted as being relatively intolerant of hydrogen sulphide. Many species are cited by (Bick, 1972) as being unable to tolerate any measurable H₂S. A few genera include species with limited tolerance, for example: Litonotus (0.5 mg/l), Euplotes (<1.0 mg/l), Tetrahymena (<2.0 mg/l), and Spathidium (no value) (Bick, 1972). These genera found in the Cayman lagoons and recovered from the lagoons in the presence of significantly higher H₂S levels (see Tables 9.6) indicate that understanding their survival in pond liquid with a wide range of physico-chemical variations remains incomplete.

Ammonia toxicity is another chemical that plays a role in species selectivity. König et al (1987) in laboratory growth studies showed that Chlorella was more tolerant than the Euglena species to ammonia at pH 9.0 and temperature 25°C (40% of ammonia in the toxic, unionised form).

It is possible for ammonia concentrations to increase in a WSP system due to microbial breakdown of urea and degradation of nitrogenous organic matter (including amino acids). Abelovich and Azov (1976), also Shillinglaw and Pieterse (1977) suggested that even at low BOD surface loadings ammonia toxicity could affect algal growth in WSP. This situation would lead to anoxic conditions and diminished performance because of the swift reduction in the algal population.
In the CI facultative ponds the predominant species, the *Merismopedia*, appears tolerant to ammonia-nitrogen levels to just over 30 mg/l, as does the *Chlorella sp* and *Euglena sp* (Fig 9.6). The protozoans such as *Chlamydomonas* and *Litonotus* were found in samples where the ammonia-nitrogen concentration was just over 10 mg/l but <15 mg/l. No *Tetrahymena*, *Euplotes*, *Spathidum*, *Epiphanes*, and *Agmenellum* were found in samples with >10 mg/l ammonia-nitrogen.

Chlorophyll *a* concentrations in the CI lagoons are lower than some reports from other studies and correlated to the low number of total cells present in each sample. This demonstrates that productivity in the CI ponds is inhibited and is probably due to the consistently high levels of H$_2$S. In the San Juan WSP system in Peru, chlorophyll *a* concentration in the range 1000-3000 µg/l is regarded as indicative of an efficiently functioning facultative pond (Pearson and Konig, 1986). They found the flagellate algal species predominate in facultative ponds (*Euglena, Chlamydomonas*) and the non-flagellate green algae dominate in maturation ponds.
In highly loaded (up to 1372 kg BOD/ha day) facultative ponds in Kenya, chlorophyll-a levels varied from 59-3178 μg/l (Alabaster et al., 1991). Pearson et al., (1987c) in Loures, Portugal documented mean chlorophyll a of 154 μg/l in a primary facultative pond; average of 1227 μg/l in secondary facultative pond; the maturation pond average was higher at 1454 μg/l. A comparison was made between grab samples and using a water column sampler. Pearson et al (1987c) found grab samples showed wider variations in concentrations throughout the day while column samples were more consistent, and concluded water column samples were a satisfactory method of collecting samples as they gave a better representation of the daily mean.

Saline waters above EC 12000-18000 μS/cm water are unfavourable to rotifers (Donner, 1966), so it is not surprising to find only one species from the Rotifera family identified in this study. The Epiphanes sp was found; which is described by Donner (1966) as common in highly polluted water such as farmyard ponds and feeds on Chlorella and Euglena. The specimens were very active and appeared to be healthy, from the keying reference (Donner, 1966) the specimens were all female.

The Epiphanes sp was observed in the final effluent from the WSP on the first 3 examinations when the sulphide level was zero. The fourth sample set examination did not reveal the presence of the species although there was no sulphide in the final effluent. However the effluent from the first maturation pond had low levels of sulphide and conceivably influenced conditions in the final pond of the treatment system. In the last examination (20 Jun91) of pond 2.2’s effluent, when the sulphide was 4.0 mg/l, no rotifers were seen. From this limited investigation it does not appear that a wide variety of rotifers are able to become established in the maturation ponds therefore their contribution to pathogen removal via predation in the saline CI system is probably limited.

From this short study it is clear that the CI wastewater ponds are limited in terms of microflora diversity compared to other investigations previously referred to. As the data generated in this investigation were limited, no further statistical evaluation was carried out. The results clearly suggest that physico-chemical characteristics (increased salinity and sulphide toxicity) and hydraulic behaviour of ponds probably have the greatest influence. Ammonia toxicity is not considered to be the most significant toxicant in the CI ponds because the pH of the ponds since 1988 rarely exceed >8.5. However further studies into the trophic interactions taking place in the complex community of normal WSP would assist in explaining the role of predation and organic matter stabilisation by zooplankton.
In developing a biotic index, it must be remembered that the behaviour of stabilisation ponds will vary with environmental/climatic variations and wastewater characteristics. Furthermore, an understanding of mechanisms operating in saline waste stabilisation ponds would undeniably augment scientific wisdom as countries with limited water resources consider alternatives to conventional wastewater treatment and look more to treatment methods incorporating reuse.

It is important to note the presence of protozoa such as *Tetrahymena* and *Euplotes* in the effluent from the maturation ponds because bacteria, including pathogenic bacteria such as those that cause cholera, typhoid, and faecal indicator bacteria such as *Escherichia coli*, constitute the main food for these ciliate protozoa (Curds and Fey, 1969). Other studies in marine environments have recognised the importance of microbial community control by protozoa grazing (Finlay *et al.*, 1988). Although the majority of research carried out regarding predation of bacteria by protozoa has come from studies in activated sludge and percolating filter sewage treatment, as Curds (1982) concluded "...the effect of predation upon bacterial activities in chemostat culture systems is ripe for investigation since the results would be applicable to all ecological situations where bacteria-feeding organisms are present".

### 9.3 Ova and Parasites - Preliminary Investigation

Although the final effluent from the Cayman WSP system was unsuitable for irrigation reuse due to high salinity, a preliminary investigation of nematode ova and parasites loading on the system was carried out on raw sewage Nov90-Feb91.

Microbiological guidelines for the use of wastewater in agriculture published by the WHO (1989), for the first time, included recommendations regarding the presence of helminth eggs. Shuval (1991) reviewed the findings of a World Bank/United Nations Development Programme that proposed the guidelines with the aim of protecting human health. He reported that a well-designed 4-5 WSP system with 20-25 days retention would be likely to achieve the required guideline of ≤1 intestinal nematode egg/l. Wastewater treatment methods such as activated sludge, oxidation ditches and aerated lagoons are not very efficient in removing helminth eggs, generally 90% (Feachem *et al.*, 1983).

Saqqar and Pescod (1991) questioned the ability of WSP, operating within low temperature range (12-15°C) and 8-10 days retention time, to effectively achieve 100% removal at all times, but did report nematode removal in the first stage (anaerobic ponds) of the Al-Samra system of 87-100% with a mean of 97%. At high temperatures
(25-27°C), the settling velocity is much faster, the beneficial effect on helminth egg removal is shown in studies carried out by Mara and Silva (1986). In pilot-scale WSP systems in Brazil, Mara and Silva (1986) reported egg free effluents after 18.9 days retention in a single primary facultative pond; 88-98% removal of Ascaris ova in anaerobic pond. They concluded that effluents with ≤1 egg/l can be produced in a 1 day anaerobic pond, followed by a 5 days retention secondary facultative pond and a maturation pond with 5 days retention. This is less conservative than the recommendation in Fieachem et al (1983) where it is stated ‘well-designed, multi-celled ponds with at total retention time >20 days’ would achieve 100% removal of helminth eggs.

In a WSP system treating wastewater produced by a small village (population 1200) in Mexico Rivera et al (1986) reported removal of Entamoeba histolytica varied between 30-100%.

The ova of parasitic intestinal worms and protozoa are removed by sedimentation because they are denser than water. Many studies indicate that by the time effluent reaches the final pond there should be no carry over of parasitic ova unless there is serious short-circuiting occurring or aerators are in use. There are, however, a number of factors that hamper sedimentation: short-circuiting, non-uniform flow, detergents, and interfering floatables (Shuval et al, 1986).

Clearly, excellent removal of helminth eggs by WSP systems is well documented. In the Cayman Islands the health status of the populace is on a par with that found in developed countries such as the USA. With a high standard of living, good water and sanitation coverage (described in Chapter 1), good hygiene (public education, schools), one would not expect to find a high prevalence of parasitic infections. In order to offer insight into helminth ova and parasite loading on the WSP the study described in this section was carried out.

9.3.1 Determination of Parasitic Ova in Raw Sewage
In order not to compromise the results due to influence of the approximately 20% final effluent recirculation and septage on the raw sewage, grab samples were collected at the main pumping station (Pumping Station # 1) on West Bay Road and at the final effluent weir chamber. Samples of 1 l volume were collected monthly during the period Oct90 to Feb91 (commencing on 26 Oct90). Because Ascaris fecundity is so high (>200000 ova released per day per worm, (Feachem et al, 1983) it is a useful indicator of the likelihood of finding human parasites at all. The detection method employed was not one of the more common methods. This was a preliminary survey to indicate whether
it was necessary to continue and expand into an in-depth study therefore the chosen method was considered appropriate. Data was obtained from the Cayman sewage and final effluent using a membrane filtration method adapted from Howard et al (1975).

The method is described as follows: 2-4 drops of Lugol's iodine was added to a 25 or 50 ml (depending on solid content) of the sample. The sample was mixed by swirling for about 15-30 seconds and then was filtered through a 0.45 μm, gridded membrane filter into a clean filtration assembly. Vacuum was applied for several minutes until the liquid was completely pulled through the membrane filter. The membrane was then transferred to a labelled petri dish and placed in a 44°C incubator for 1-2 hours until completely dry. To facilitate the microscopic examination, the membrane was cut along the grid lines (approximately 8 x 8 squares) and mounted on a slide in immersion oil with care to avoid air bubbles. The membrane became transparent, sufficient immersion oil (1-2 drops) was added to the top of the membrane in order to float a cover slip over the membrane. The entire area was scanned at x100 magnification, suspicious egg shapes were viewed at x400 to confirm identity. The number of eggs were recorded per square and converted to count/l. The above procedure was repeated 5 times on each 1 l sample and the mean number of eggs/l calculated. The inadequacies of this method are recognised and results are regarded as an approximate first estimate. Nonetheless the counts compared closely with those found in a survey by Ellis et al (1993).

9.3.2 Results of Survey
Preliminary surveys showed an average range of 5-15 Ascaris eggs/l with individual counts from 0-50 Ascaris ova/l in the raw sewage samples. The data confirmed, as suspected, that the prevalence of the Ascaris parasite in the largely tourist human population, for the area sewered, was very low. Since the raw sewage counts are low it was not considered possible to get accurate and useful information on removal rates through the sewage lagoons. No Ascaris ova were detected in the final effluent samples. From the data it may be assumed that the system approaches 100% removal efficiency and this investigation was subsequently terminated. Nominal retention through the system during this period averaged 21 days which is within the range reported to produce effluent free of parasites (Arceivala et al, 1970; and Shuval et al, 1986). Studies on the San Juan lagoons in Peru indicated 100% removal of helminths was achieved in the 2 treatment ponds in series (Yanez et al, 1980) with 5.5 days retention, however enteric protozoa removal required 2 ponds in series with a total retention time of 36.7 days. Dixo et al (in press) reported studies in Brazil on a series of WSP with cumulative retention time of 20 days were effective in removing high levels of intestinal nematode eggs.
Data obtained from Medical Laboratory (CI Health Services, 1994) at the George Town Hospital, on the occurrence of various parasitic diseases in Cayman in 1994 is presented in Table 9.7 and illustrated in Fig 9.7.

Table 9.7 Enteric parasites and ova cases reported by the George Town Hospital, Grand Cayman in 1994 and the estimated prevalence per thousand population.

<table>
<thead>
<tr>
<th>Parasites and Ova</th>
<th>Cases reported</th>
<th>Prevalence/1000 population</th>
</tr>
</thead>
<tbody>
<tr>
<td>Entamoeba histolytica</td>
<td>2</td>
<td>0.1</td>
</tr>
<tr>
<td>Ascaris lumbricoides</td>
<td>5</td>
<td>0.17</td>
</tr>
<tr>
<td>Trichuris trichiura</td>
<td>9</td>
<td>0.3</td>
</tr>
<tr>
<td>Giardia lambia</td>
<td>44</td>
<td>1.5</td>
</tr>
<tr>
<td>Entamoeba coli</td>
<td>30</td>
<td>1.0</td>
</tr>
<tr>
<td>Strongyloides</td>
<td>2</td>
<td>0.1</td>
</tr>
<tr>
<td>Necator americanus</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Ancylostoma</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td><strong>92</strong></td>
<td><strong>3.1/1000 population</strong></td>
</tr>
</tbody>
</table>

The data in Table 6.7 do not represent every single case occurring on Grand Cayman in 1994 as there is another medical laboratory which is privately owned and the author did not attempt to obtain data from there.

From the Fig 9.7 it is clear that *G. lamblia* is the enteric parasite found most frequently in the population of Grand Cayman followed by *Entamoeba coli*. One of the most commonly identified human enteric pathogens in the United States is the *Giardia sp* parasite. Conventional treatment plants are not as effective as WSP in removing parasitic ova. Some studies point out the difficulties and discrepancies in recovery methods Wiandt *et al* (*in press*), they also report that coprological studies indicate that *Giardia* is present in 6% of the population of southern France and that raw sewage containing on average $0.23-25 \times 10^3$ cysts upon treatment in a WSP system with 30-40 days retention time, attained 99.78-100% removal.
In Ellis et al., (1993) results are presented from an 18 month study of parasite ova and cysts presence and their removal in the Cayman WSP. Considering the data in Table 9.7, it is difficult to reconcile the high counts of *Necator americanus* they reported in the raw sewage and final effluent from pond 2.2 with prevalence of this parasite in Grand Cayman. Ellis et al. (1993) agree that parasitic infestation is not endemic in the Grand Cayman population however they did not present data to estimate prevalence. Additionally the study failed to critically evaluate data reporting 33-690 *N. americanus* eggs/l in the final effluent of pond 2.2. It seems highly unlikely that such levels would be found in the raw sewage and even less so in the final maturation pond.

Although, Ellis et al. (1993) hypothesized that septage is the likely source of the reported high concentrations of enteric helminth ova/l, it should be first considered that septage in the Cayman WSP represents only <1% of the average daily flow. The report did not explain whether samples were collected during the simultaneous emptying of septage tankers. With the obvious dilution into the ponds, it is difficult to explain a
count of 690 *N. americanus* ova/l in the final effluent after a nominal retention time of 23 days. This would indicate that the initial loading was unrealistically high or that *Necator* has been misidentified. While Ellis *et al* (1993) discounted short-circuiting in the WSP system, retention time studies presented in Chapter 10 reveal that short-circuiting indeed occurs to a large extent in the CI ponds. Thus although there may be some difficulties with the numbers reported, short-circuiting identified may be the cause of the positive results in the maturation ponds. It is certain that sludge from septic tanks will contain high levels of helminth eggs but the mere fact of dilution raises concerns about the numbers reported.

Table 9.8 Reported numbers of *Ascaris* ova in raw sewage
(some references from Feachem *et al*, 1983).

<table>
<thead>
<tr>
<th>Author/s</th>
<th>Place</th>
<th><em>Ascaris</em> ova/litre</th>
<th>Prevalence Rate %</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bhaskaran (1956)</td>
<td>Calcutta, India</td>
<td>200-2130</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Aleppo, Syria</td>
<td>1000-8000</td>
<td>42</td>
</tr>
<tr>
<td>Wang and Dunlop (1954)</td>
<td>Colorado, USA</td>
<td>5-110</td>
<td></td>
</tr>
<tr>
<td>Frederick (this study)</td>
<td>Cayman Islands</td>
<td>5-15</td>
<td>&lt;1% (0.17 per thousand)</td>
</tr>
<tr>
<td>Nupen and de Villiers (1975)</td>
<td>Daspoort South Africa</td>
<td>19</td>
<td></td>
</tr>
<tr>
<td>Kalbe (1956)</td>
<td>Germany, GDR</td>
<td>&lt;3&lt;sup&gt;8&lt;/sup&gt;</td>
<td></td>
</tr>
<tr>
<td>Liebmann, (1965)</td>
<td>Tokyo, Japan</td>
<td>10-80</td>
<td></td>
</tr>
<tr>
<td>Grimason <em>et al</em> (<em>in press</em>)</td>
<td>Kenya</td>
<td>17.5-133.5</td>
<td>8.2-82</td>
</tr>
<tr>
<td>Rowan and Gram (1959)</td>
<td>San Juan, Puerto Rico</td>
<td>14-38</td>
<td></td>
</tr>
</tbody>
</table>

In addition to the data reported in Table 9.8, there are wide variations in helminths eggs/l in raw sewage: for example; 219 eggs/l reported by Grabow and Nupen, (1972); 450 eggs/l reported by Panicker and Krishnamoorthi (1981); and 2330 eggs/l was reported by Fitzgerald (1977). In Marrakech, Morocco, Schwartzbrod *et al* (1987) reported an average of 11.7 eggs/l in the incoming sewage which was treated by a two pond lagoon system with 8-39 days retention time, producing effluents completely free of helminth eggs during the study period of 1 year. However they reported that the sludge contained 75-200 eggs/l.

From this preliminary investigation it was concluded that:

- the prevalence of infections in Grand Cayman due to enteric parasitic pathogens was low (3.1 per 1000 population);
• because of the socioeconomic status of residents in the sewerage drainage area the sewage had correspondingly low concentrations of helminth ova (5-15 *Ascaris* ova/1);
• reuse of the final effluent for irrigation purposes was not a choice therefore no direct health risk; and
• disposal of the final effluent through deep wells did not present a risk to humans in the environment.

In consideration of the above it was decided to terminate the ova and parasite investigation until the need for further data arose.

### 9.4 Pathogen Indicator Bacteria Die-off

Waste stabilization ponds are considered to be the most efficient low-cost method of sewage treatment for pathogen removal. The biological and physical nature of a pathogenic organism will affect the ability of a pond to remove it. Pathogens are categorised into three broad groups; helminths and protozoa, viruses, and bacteria.

The fate of pathogenic organisms in WSP systems is of major concern due to public and environmental health considerations. It is especially important if reuse is considered a discharge option. Bacteria, protozoa, viruses, parasitic ova and fungi are some of the organisms found in wastewaters that can cause infectious diseases (Joshi *et al*, 1973; Kott, 1984; Rose, 1988; McFeters and Singh, 1991; Saqqar and Pescod, 1991; and Wolfe, 1992).

It is not routine to determine each pathogen specifically as this would involve time consuming and complex procedures, consequently, indicator organisms are used. The intestinal tract of human and animals is the natural habitat of enteric bacteria, some of which are pathogenic. The following characteristics define the ideal faecal indicator bacterium (Feachem *et al*, 1983):

• a natural and exclusive inhabitant of the intestinal flora of healthy humans and animals;
• non pathogenic and present in higher numbers than faecal pathogens;
• equal or more resistant than enteric pathogens to disinfectants and environmental stress;
• the method to recover and count is simple, cost effective, reliable and reproducible;
• detectable in small numbers and with precision;
• unable to reproduce outside the intestines; and
• has a decay rate slightly less than that of faecal pathogens.

There are few organisms that are able to satisfy all these requirements in water. The classical indicator bacteria used are the coliforms (total and faecal coliforms groups, the faecal streptococci and Clostridium perfringens).

The most commonly used indicator for wastewaters is the faecal coliform group. Because the faecal indicator bacteria are commensal, non-pathogenic and are present in high numbers in human and warm, blooded animal excreta, their presence in potable water or wastewater is considered a reasonable indicator that the water has been contaminated with excreta.

The faecal coliforms and faecal streptococci bacteria, in spite of their inadequacies, are utilised because they are always present in high numbers and some biotypes of E. Coli are the commonest cause of diarrhoea including travellers' diarrhoea. Thus they remain the most widely used faecal indicator bacteria for the indication of the pathogen content and for the evaluation of the quality of treated wastewater effluents.

9.4.1 Indicator Bacteria Removal in WSP
As with drinking water quality, the major indicator for the removal of pathogens in a WSP system is faecal coliform bacteria. Much has been written in the relevant literature concerning faecal coliform removals and the subsequent assumption of pathogen reductions. Faecal coliform removals have been attributed to several factors: low nutrients and competition; redox potential; pH; sunlight; dissolved oxygen; short-circuiting/retention times; algal derived chemical changes; and protozoal predation.

There is much speculation in the literature concerning the excellent removal of faecal coliforms by sewage lagoons. Experiments have been performed which show that a combination of high pH, sunlight, oxygen levels correlate with good faecal coliform removals.

Starvation due to the pond environment being low in nutrients has been assessed and it is shown that the bacterial cell will adapt itself to lower nutrient concentrations after a few hours of exposure to those conditions. The organism will survive as long as it can scavenge exogenous material and draw on its endogenous reserves.
Retention times have been repeatedly discussed in pond literature as providing the time for the other mechanisms to remove indicator bacteria. If short-circuiting occurs, a fraction of the sewage will pass through the system with barely any exposure to removal mechanisms, thus dramatically reducing the ponds' efficiency. In order to assess whether short-circuiting is occurring to a significant extent in the Cayman ponds, a tracer study was undertaken using bacteriophage tracers (described in Chapter 10).

9.4.1.1 Faecal Coliform Die-off Study
Many observations and studies have been carried out on the % removals in ponds of the indicator bacteria, faecal coliforms - both experimental and field. Some researchers question the reported excellent indicator bacteria removal because the retention times are longer than in conventional treatment therefore they feel that WSP treatment of sewage is inefficient (James, 1987). However this is not the general opinion, and the author does submit that the removal of pathogen indicators in the CI ponds is good in spite of operational problems with hydraulic overload and salinity (Chapter 7).

A large number of theories have been suggested to describe the mechanisms involved in removing pathogenic bacteria from ponds some of which are shown in Table 9.9:

Table 9.9 Faecal indicator bacteria removal mechanisms.

<table>
<thead>
<tr>
<th>REMOVAL MECHANISM</th>
<th>REFERENCES</th>
</tr>
</thead>
<tbody>
<tr>
<td>Algal produced toxins</td>
<td>Pratt, 1944</td>
</tr>
<tr>
<td>Bactericidal effect of sunlight</td>
<td>Pratt, 1944; Gameson and Gould, 1985;</td>
</tr>
<tr>
<td>High pH</td>
<td>Pratt, 1944</td>
</tr>
<tr>
<td>Photooxidation</td>
<td>Curtis, 1990</td>
</tr>
<tr>
<td>Predation</td>
<td></td>
</tr>
<tr>
<td>Sedimentation</td>
<td></td>
</tr>
</tbody>
</table>

Some of the main factors that influence faecal coliform death and survival in facultative and maturation ponds are shown in Table 9.10:
Table 9.10 Some of the mechanisms affecting the faecal coliform death and removal in the major types of waste stabilisation ponds.

<table>
<thead>
<tr>
<th>ANAEROBIC</th>
<th>FACULTATIVE</th>
<th>MATURATION</th>
</tr>
</thead>
<tbody>
<tr>
<td>sedimentation</td>
<td>dilution</td>
<td>algal toxicity, high pH, photooxidation</td>
</tr>
<tr>
<td>retention time, short circuiting</td>
<td></td>
<td>predators</td>
</tr>
<tr>
<td>temperature</td>
<td></td>
<td>starvation</td>
</tr>
<tr>
<td>sedimentation</td>
<td></td>
<td>dilution</td>
</tr>
<tr>
<td>retention time</td>
<td></td>
<td>retention time</td>
</tr>
<tr>
<td>temperature</td>
<td></td>
<td>temperature</td>
</tr>
</tbody>
</table>

Throughout the literature reviewed a confusing array of k values were reported for faecal coliform die-off in ponds. Assuming that complete mixing occurs then the faecal coliform removal follows first-order reaction kinetics (Chapter 2, Eq 2.5). However under batch or plug-flow conditions, bacterial removal is described by Chick’s Law:

\[ \frac{N_t}{N_i} = 10^{\frac{t}{t_b}} \]

Eq 9.1

Where

- \( N_t \) = number of faecal coliforms /100 ml100ml, at time t
- \( N_i \) = number of faecal coliforms /100 ml100ml, initial population
- \( k_b \) = first-order faecal coliform removal constant (d\(^{-1}\), log\(_{10}\)
- \( t \) = mean retention time in pond, days

At the design stage, one of the difficulties faced is the assigning of a value to the reaction rate ‘constant’, k. The k value depends on factors such as sewage characteristics, pond and air temperature, retention time, climatic conditions, physical characteristics of the pond. In determining reaction constant values for faecal coliform removal efficiency, Marais (1970) found that temperature was the most important factor. His work led to the development of the formula commonly used to determine a k value in the design of maturation ponds (Chapter 5, Eq 2.5).

One of the difficulties in the assignment of a k value to a WSP system is that it assumes that the reaction-rate constant is the same for all maturation ponds in that system. The reaction-rate constant in each pond will differ considerably because the density of faecal
coliforms in the influent to each pond in a series will be different. Even in systems operated in parallel, $k$ values for ponds at the same stage of treatment may vary significantly due to variations in the hydraulics or other inexplicable factors.

However, some experts (Yanez, 1993) still consider that it is essential to determine $k_b$ values. The values provide a basis for comparison with the actual reductions provided by the monitoring programme in addition to assisting in the development of further designs (under similar conditions). If the $k_b$ is significantly different from the $k$ demonstrated in the monitoring programme then something may be fundamentally wrong with hydraulic and retention time in that system component.

The objectives of this investigative study were to determine the $k_b$ values (the batch decay coefficient for faecal coliform die-off in day$^{-1}$) at the maturation pond stage in the Cayman WSP. The Cayman maturation ponds were designed based on $k$ values calculated using the ambient temperature 24°C.

In order to determine the $k_b$ values for the CI WSP system, the experiment described as following was carried out in batches in containers suspended in the lagoons.

**9.4.2 Experimental Method**

This experiment was carried out using pond effluent from facultative pond 1.1 and from maturation pond 2.1. Two 40 l containers were lined with black garbage bags (simulating the lining of ponds). The containers were then each inserted into a rubber inner tyre tube in order to suspend/float the container in their respective pond.

One container was filled with pond effluent from the overflow weir of pond 1.1 which was then moored by a nylon rope to the outlet point of that pond (Fig 10.8 and 10.9). Additionally a maximum/minimum thermometer was suspended midway into the pond liquid in the container. The same procedure was repeated for the effluent from maturation pond 2.1.

The sample taken at the outlet of pond 1.1 represented primary settled sewage, while that of the maturation pond 2.1 represented what would be treated in maturation pond 2.2.

The first experiment (Exp 1) was carried out in Jan93 for 10 days and operated with a liquid temperature range of 23.5-26.0°C. A second experiment (Exp 2) was carried out in Feb93 however not all chemical parameters were analysed due to manpower restraints. Exp 2 was extended to 11 days and operated with a liquid temperature range
of 27.0-29.0°C. The third experiment (Exp 3) run was carried out in Mar93 for 14 days during which the liquid temperature ranged from 27.0-34.0°C.

All samples were collected daily for the first 4 days and then every other day up to the 14th day in the case of Exp 3.

The samples taken were analysed for pH, EC, SO₄, H₂S, suspended solids, and faecal coliform bacteria. On site measurements were DO (Exp 1 and 3, only), EC, and °C. A record was made of the physical appearance of the liquid in each container in addition to the daily rainfall measurements.

9.4.3 Results
The results from the experiments are shown in Table 9.11.
Fig 9.8 Container for faecal coliform die-off experiment being filled with effluent from facultative pond 1.1

Fig 9.9 Faecal coliform die-off batch experiment showing measurement of max/min temperature, the floating device and anchoring strategy.
9.4.4 Discussion

Conventional wisdom is that anaerobic and facultative ponds are designed to remove BOD most efficiently whereas maturation ponds are more efficient at removing pathogens and faecal indicator bacteria. This has not proved to be the case in the exceptional circumstances prevailing in the Grand Cayman in Phase 1 (Table 5.2). The
facultative ponds achieved 99-99.9% removals of faecal coliforms whilst each of the two maturation ponds in series could only add 76-88% removal. Thus the maturation ponds are clearly identified as failing to perform adequately with respect to faecal coliform (FC) removal. In Phase 2 (Table 6.6) maturation pond performance declined further and each of the two maturation ponds added only 62-81% removals.

The key question addressed here is whether the performance of maturation ponds is being compromised by hydraulic overload, biochemically adverse conditions or a mixture of both. The additional question is posed as to whether various decay rate constants can shed any light on what mechanisms are primarily responsible for impaired performance.

The evidence from Phase 1 is not that helpful. In Phase 1 there were only 4 months (in the first year) when the FC standard was consistently achieved, even though the flow was then within the design flow. The fact that the FC loading rose steadily, from $10^6$ cfu/100 ml at the start of Phase 1 to $10^9$ cfu/100 ml towards the end of the period, was a confounding variable whilst salinity and flow also continued to rise.

With a worsening performance in the maturation ponds in Phase 2 it seemed logical to investigate the FC die-off in batch experiments and compare their removal with that demonstrated in the Phase 2 routine monitoring programme. It was argued that the decay rate constants calculated from both the batch experiments and empirical monitoring programme should also be compared with the constant of Marais used in the original design of the ponds.

If the batch removal results were comparatively poor and the batch decay rate constants were low, compared with the design k values, then it may be deduced that the poor removal is linked to biochemical imbalance due to salinity-associated hydrogen sulphide. If however batch removal is better than, or comparable to the monitoring programme results, then it is likely that the problem in the maturation problem is hydraulic.

The batch FC removal results for the maturation ponds in Jan93 demonstrate a 4.2 log removal in 6 days. In Mar/Apr93, 4.2 logs were removed in 8 days. The decay graphs are presented in Figs 9.10-9.15.
The monitoring programme shows that facultative pond 2.1 produces less than an overall 80% FC reduction throughout the Phase 2 period 1990-94. This is reflected in a decay rate constant of only 1.93 compared with the Marais design constant of 6.2 at 24.9°C (Table 9.12).

Reduction in Pond 2.2 is even worse with an average for the same period of only 67% and a decay rate constant of 1.43. Both results suggest substantially reduced retention times which fit with the reduced volume available due to sludge deposition, and with the excess flow received.
Table 9.12 Comparison of faecal coliform decay rate constants (k) for maturation ponds.

<table>
<thead>
<tr>
<th></th>
<th>Pond 2.1</th>
<th>Pond 2.2</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>A. Monitoring programme</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Jan93</td>
<td>2.27</td>
<td>0.59</td>
</tr>
<tr>
<td>Feb93</td>
<td>1.52</td>
<td>0.61</td>
</tr>
<tr>
<td>Mar93</td>
<td>2.26</td>
<td>1.70</td>
</tr>
<tr>
<td>Apr93</td>
<td>2.27</td>
<td>0.78</td>
</tr>
<tr>
<td>Mean</td>
<td>2.08</td>
<td>0.81</td>
</tr>
<tr>
<td>Mean for Jun90-Dec 94</td>
<td>1.93</td>
<td>1.43</td>
</tr>
<tr>
<td><strong>B. Batch die-off study</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Jan93</td>
<td>2.08</td>
<td>2.06</td>
</tr>
<tr>
<td>Feb93</td>
<td>2.29</td>
<td>0.74</td>
</tr>
<tr>
<td>Mar93</td>
<td>1.63</td>
<td>1.88</td>
</tr>
<tr>
<td>Mean</td>
<td>2.00</td>
<td>1.54</td>
</tr>
<tr>
<td><strong>C. Marais design constant at 25°C</strong></td>
<td>6.2</td>
<td>6.2</td>
</tr>
</tbody>
</table>

It is therefore of particular interest that the decay rates for pond 2.1 calculated from the 3 batch experiments in the period Jan93 to Apr93 produce an average k value of 2.00, this compares with a mean value of k = 2.08 for the routine monitoring programme for the same period and an average of 1.93 for the entire period from 1990-Dec94 inclusive. This close agreement suggests that part of the problem is indeed hydraulic in facultative pond 2.1 whereas the very major discrepancy with the Marais design k of 6.2 suggests that the anoxic, sulphide-rich conditions are also contributing to and compounding the problem.

In the case of maturation pond 2.2, the even lower removal is reflected in the correspondingly lower k values in the batch study. The average k batch value of 1.54, compares poorly with a mean value of k = 0.81 for the routine monitoring programme for the same period but rather better with the average of 1.43 for the entire period from 1990-Dec94 inclusive.

9.5 Summary of Investigative Studies
The 3 short-term studies previously described investigated some of the ecological mechanisms operating in ponds and their effect on pathogen removal. The main conclusions from each study are summarised below:
1. **Functional Ecology**

The diversity of beneficial microbiota in ponds is severely limited by excessive hydrogen sulphide production. The variety of species in the pond are dominated by the cyanophyta - *Merismopedia sp*. The ponds however are still able to reduce faecal coliform indicator bacteria by a mean of 3 logs.

2. **Ova Parasite Study**

This investigation demonstrated that raw sewage entering the sewage treatment works had low levels of *Ascaris* ova, 5-15 eggs/l while none were detected in the final effluent from pond 2.2. The study was discontinued as it was clear that the low rate of prevalence (3.1 cases per 1000 population) was unlikely to result in sufficient concentrations of parasites to permit a meaningful evaluation of performance through the WSP system.

3. **Pathogen Indicator Bacteria Die-off Study**

From this study the k values calculated from the monitoring programme compared favourably with the values obtained in the batch study however both were significantly lower than that expected using the Marais formula. This clearly demonstrates the inherent weakness of using k values based on temperature alone, when in reality uncontrollable or unavoidable variables such as climatic conditions or saline sewage confound the removal/die-off of bacteria in WSP systems.
CHAPTER 10

10.0 RETENTION TIME INVESTIGATIONS USING SERRATIA MARCESCENS BACTERIOPHAGE AS A HYDRAULIC TRACER

10.1 Introduction
The monitoring results of the WSP under study show that the ponds are not achieving the performance predicted from the design assumptions. There are several factors that could be contributing to this poor performance; biochemical malfunctioning due to salinity levels (discussed in Chapter 8), dilution due to excess non-sewage water entering the system (discussed in Chapters 6-7) causing hydraulic overload, and short-circuiting.

It is clear that the hydraulic characteristics of a wastewater pond treatment system have considerable and overriding influence on treatment efficiency. Severe short-circuiting will result in poor treatment as the percentage of the wastewater that passes out as effluent, before it has had time to benefit from the natural purification processes, will retain high concentrations of pathogens. In biochemical reactors such as waste stabilisation ponds, BOD, settlement and pathogen removal efficiencies are all affected by the mixing characteristics of the regime.

Factors that contribute to the hydraulic behaviour of WSPs are environmental conditions such as temperature, windspeed and direction, in addition to physical properties such as shape, inlet and outlet juxtaposition. The logical approach to describe the mass pollutant transport and design for their removal in biochemical reactors such as waste stabilisation ponds is to combine equations of continuity and motion (basic hydrodynamic laws) with the methods of chemical reactor design.

It follows that for a more complete understanding of the performance efficiency of a treatment system, an evaluation of its hydraulic regime is vital. Being able to measure the mixing characteristics of the treatment system and correctly interpret the results will greatly assist in the prediction of treatment efficiency and pond behaviour under similar and different operating conditions. Verification of the design constants for residence time, BOD and faecal coliform removal should also be possible.
Finney and Middlebrooks (1980) carried out an evaluation of equations used in the design of facultative ponds. These design methods are based on organic and hydraulic detention times; empirical design equations; and, rational design equations (Chapter 2).

The various equations available to engineers for the design of ponds are based on differing mixing assumptions ranging from plug flow to completely mixed reactors. These assumptions have been and still are used by waste stabilisation pond scientists and engineers to describe the hydraulic behaviour in ponds.

10.2 Retention Time in WSP
In order to completely evaluate the ponds' performance a study of the mean hydraulic retention time and circulation patterns will be required. This will assist in determining the existence and the extent of short circuiting and the percentage of dead volume. Information obtained from a properly designed tracer study may then be used to improve inlet and outlet arrangements of the ponds and other design features.

The consistent prediction of pond performance by any design method without accurate projections of hydraulic residence time is impossible. It is recommended that the effect of physical configuration and climatic conditions on hydraulic residence be carefully considered. Some investigators (Neel et al, 1961) have suggested that the factor having the most affect on reaction rates is hydraulic retention time. Although hydraulic flow characteristic and its relation to pond performance and effluent quality have received some attention from various studies, it has most often been paid little attention or overlooked in the design stage (Marecos do Monte and Mara, 1987; and Juanico, 1991).

Attempting to describe the hydraulic behaviour in ponds is not a simple matter and in order to do this a tracer study must be carried out. In tracer studies care must be taken to minimise the large analytical and sampling errors inevitable with the unavoidable large dilution of the tracer. Although there have been some investigations in the hydraulic regime of ponds, they are not great in number.

The opportunity to use the CI full-scale pond system as experimental ponds should not be ignored. As there is a reasonable amount of flexibility in terms of flow arrangements, parallel systems could be compared and evaluated. Tracer studies using these experimental arrangements could be of tremendous value to the Authority as decisions regarding the present and future designs would have sound data on which to base them.
In many cases, design models have been proposed based on laboratory-scale ponds where different types of flow reactors can be closely controlled. In fact Wood (1987) poses the question as to what extent a laboratory-scale system can ever be representative of the conditions in a large-scale WSP. He pointed to instances where continuous plug-flow models were used to determine reaction rates when in the laboratory-scale experiment the reactors were fed semicontinuously. Significantly different reaction rates were obtained when he analysed the data using a semicontinuous model.

10.3 Hydraulic Flow Patterns
Hydraulic flow patterns in most lagoons fall between the two extremes of plug flow or completely mixed. In between there is the dispersed flow (non-ideal) pattern. Definitions of flow patterns were developed by chemical engineers for chemical reactors and have been adopted for and adapted to waste stabilisation pond design by sanitary engineers (Thirumurthi, 1969). In the design of ponds, hydraulic behaviour is usually based on the assumption of plug-flow or completely-mixed flow, and less frequently on dispersed or non-ideal flow models. Consequently, removal efficiencies in WSP systems are based on equations related to each particular flow model and yet in most situations non-ideal flow prevails. Juanico (1991) developed mathematical models simulating plug-flow and perfect-mixed systems to evaluate the effect of flow pattern on the performance of WSP. Others such as Agunwamba et al (1992) have also developed mathematical models to predict dispersion numbers (d) for non-ideal or dispersed flow model.

10.3.1 Plug Flow
Plug flow or piston flow as it is sometimes referred to, may be visualised as each plug of sewage enters the reactor it does not mix with the older contents but passes along the length of the reactor behind the previous plug until it reaches the outlet (Fig 10.1). Consequently, every element in the reactor is treated for the same amount of time (retention time).

Reactors that exhibit plug flow are characterised by the following equation for first order chemical reactions:

\[ \frac{C_i}{C_0} = e^{-u} \]  
Eq 10.1
Several studies have proposed that plug-flow models are more efficient than perfect-mixed ones in terms of bacterial removal (Polprasert *et al.*, 1983; and James, 1987) but others have pointed out that predictions from plug-flow models are too optimistic in terms of efficiency (Moreno, 1990; and Saenz, 1988) and it is impossible to achieve in practice (Thackston *et al.*, 1987).

By using a series of completely mixed reactors, a hydraulic regime close to plug flow may be attained. Because removal efficiencies are greater in a plug-flow system, a series of small ponds will be more effective than a single large pond.

Due to the effects of wind induced mixing, flow turbulence and short-circuiting, plug flow conditions in a sewage treatment lagoon are not expected and are probably rare. However, an entire WSP system consisting of many ponds in series may approach plug flow behaviour. It has been suggested that the design of facultative and maturation ponds should favour plug flow over complete mixing. Pearson *et al.* (*in press b*) from research studies carried out in Brazil on experimental systems, suggest that a length to width ratio of up to 6:1 has little impact on pond performance and effluent quality. This contrasts with a
previous report from some of the same authors where they concluded that maturation ponds should be designed for plug flow hydraulic regime and reduced wind effect (Pearson et al, 1988). This suggests that even after the 30 plus years of major research that understanding and explaining some of the basic as well as the complex and dynamic behaviour of ponds continues to be an elusive goal.

10.3.2 Completely Mixed Flow

Complete and instantaneous mixing of influent with the entire contents characterises the completely mixed reactor (Fig 10.2). This results in an effluent that is identical to the contents of the reactor. The following formula represents removal efficiency in a completely mixed flow reactor:

\[
\frac{C_e}{C_i} = \frac{1}{1 + k \tau}
\]

Eq 10.2

Thirumurthi (1974) in contrast to the assumptions on hydraulic flow made by Marais (1970), takes the position that a completely mixed-flow model should never be used for the rational design of ponds because in the real world, complete mixing can never be achieved in large reactors. This has important implications for pathogen removal where the exit of a very small percentage of pathogenic microorganisms may present a substantial risk in the receiving environment.
Some researchers present confusing arguments and conclusions regarding the most effective flow regime for pathogen or substrate removal efficiencies. For instance, James (1987), presents an argument where he shows that a plug-flow regime (in the anaerobic pond stage) would improve WSP bacterial removal efficiencies from <90 to 99.999. He also suggests that in the facultative and maturation stage, a depth of 1 m would be beneficial in increasing bacterial removal due to increased exposure to light and ‘maximised mixing due to wind action which would reduce any tendency to short-circuiting!’

As previously noted and like many other flow systems, neither plug flow or completely mixed flow actually exists in WSP, normally there is a combination of both which is termed non-ideal or dispersed flow.

10.3.4 Non-ideal or Dispersed Flow

The non-ideal flow or dispersed flow model has become accepted as more representative of the flow characteristics of waste stabilisation ponds than either plug flow or completely mixed models. To describe the combination of the plug flow and completely mixed flow reactors, Wehner and Wilhelm (1956) proposed the following equation to determine removal efficiencies:

\[
\frac{C_r}{C_i} = \frac{4a \exp(1/2d)}{(1 + a)^2 \exp(a/2d) - (1 - a)^2 \exp(-a/2d)}
\]

Eq 10.3

where

- \( a = 1 + 4ktd \)
- \( d \) = dispersion number = \( D/uL \) (dimensionless)
- \( D \) = axial-dispersion coefficient (\( L^2/T = m^2/hour \))
- \( u \) = fluid velocity, (\( L/T = m/hour \)), and
- \( L \) = characteristic length (L)

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Dispersion numbers or factors, identify the type of flow, zero being characteristic of plug flow reactors and infinity of completely mixed flow reactor. In mechanically mixed reactors such as activated sludge the dispersion numbers, $d$ may range from 4.0 to infinity (Dillaha et al, 1986; and Metcalf and Eddy, 1972). Sewage treatment ponds have values of $d$ that generally fall between 0.1-2.0 (Metcalf and Eddy, 1972).

Thirumurthi (1969) recommends that the Wehner-Wilhelm (1956) equation for dispersed/non-ideal (arbitrary) flow (Eq 10.3) be used for facultative pond design because their flow patterns are more similar to plug flow rather than completely mixed flow. He developed a design formula chart for facultative ponds. In this chart, $kt$ is plotted against $C/C_i$ for dispersion factors ranging from zero (0) to infinity ($\infty$).

This chart, although useful, is limited because it requires knowledge of the dispersion factor and the $k$ value. These values even when obtained by tracer studies and batch experiments under local conditions will have a wide variation due to the many factors that affect them (temperature, wastewater characteristics, algal activity, etc.). Therefore most pond designs assume complete mixing as recommended by Marais and Shaw (1961) because it allows a margin of safety in that the retention time required for dispersed flow is less than that under completely mixed conditions.

10.4 Investigating Retention Time - Tools
The retention time is an important tool in evaluating the efficiency of a pond system because, in all cases treatment depends on retention time to some degree. The hydraulic behaviour in ponds may be influenced by a wide variety of controllable (in design) and
uncontrollable factors such as climate. The windspeed, wind direction, inlet and outlet effects, shear stresses at the sides and bottom, and buildup of sediment are some of the factors that affect the hydraulic efficiency of shallow, outdoor basins (Thackston et al, 1987).

The degree of mixing, retention time and short-circuiting in a reactor may be investigated using tracers introduced at the inlet, tracked within the reactors (ponds) and measured at the outlets as a function of time. A tracer test is used to determine the dispersion and mean retention time in a reactor. Ferrara and Harleman (1981) however, consider that even if dye tracer studies are carried out for a pond that the results are only applicable to that particular pond under the same exact conditions during which the study was carried out. Nevertheless, understanding removal efficiencies in terms of pond performance as well as illustrating design inadequacies are some of the benefits of conducting a tracer study.

The ideal tracer must be:

1. Easily and precisely detected at low concentrations.
2. Non-toxic to the investigator and receiving environment.
3. Not degraded within the reactor, or its degree of degradation known.
4. Not bound to the reactor contents.

Among the types of tracers used in wastewater treatment systems or other aquatic environments are lithium tracers (Price et al, 1973), floats (eg oranges), chemical salts, radioisotopes, fluorescent (sulphorhodamine B) and nonfluorescent dyes (Marecos do Monte and Mara, 1987). Sodium chloride was not appropriate for the CI ponds due to high concentrations (5000 mg/l) in the raw sewage.

The use of conventional tracers such as dyes, radioactive material and lithium chloride were briefly considered. However, the disadvantages of expense and potential health hazards, resulted in the search for an alternative tracer. It was decided to investigate the suitability of a non-pathogenic microbial tracer which is environmentally rare, is simple, rapid and inexpensive to produce and recover.

Biological tracers such as yeast, bacteria and phage have been extensively reviewed by Keswick (1982). Yeasts are considered the least sensitive because they are very large (1000 nm) and are naturally present in high numbers in polluted waters. Some bacteria that have been used as a tracer are capable of reproducing in the environment; potential problem with releasing antibiotic resistant trains into the natural environment.
As the bacteriophage are not known to cause diseases in humans, animals, or plants unless the host is available. From Keswick (1982) bacteriophage are less susceptible to removal by adsorption and has a density close to water - considered as potential model of human enteric viral behaviour.

Bacteriophage was the tracer medium chosen for the Cayman WSP system. Phages are virus-like particles that use or rather infect specific bacteria as the host organism and pose no human health risk at all. They consist of a nucleic acid molecule called a genome and are surrounded by a protein coat, the capsid (Tikhonenko, 1970). The process of infecting the bacterial host cell, similar to that of a pathogenic virus infecting a human cell, occurs in 5 stages:

1. adsorption to the host cell
2. ejection and penetration of the genome
3. synthesis of phage micronucleus
4. assembly of mature phage particles
5. lysis of host cell and release of phage replicates

Bacteriophages have been used experimentally for tracing groundwater migration (Bitton and Gerba, 1984; and Skilton and Wheeler, 1988), to study dispersion flows in sea-outfalls and river, and in drinking water and conventional wastewater treatment systems as a model for virus removal (Wimpenny, 1977). In the literature reviewed, only one reference to the application of bacteriophage (specifically the Serratia phage) as a tracer in WSP systems was found (Mara and Pearson, 1986).

There are a number of advantages to using bacteriophage as a tracer:

- good survival
- can be cultured to very high concentrations allowing dilution rates up to $10^{11}$
- simplicity, sensitivity and low cost of detection method
- rapid growth
- long storage life
- easy identification and numeration
- harmless to humans and environment
Some of the disadvantages to using bacteriophage tracers in wastewater treatment ponds are:

- adsorption to suspended particles
- possible inactivation by harsh environment during sewage treatment
- re-introduction to pond due to recirculation of final effluent as in the Cayman WSP system operation.

Other probable difficulties such as the presence of background organisms can be addressed by using phages of bacterial species such as a *Serratia* and *Erwina*, which are not commonly found in the environment. In the selection of a specific phage, evaluation of its survival behaviour in adverse conditions is the main consideration.

10.4.1 *Serratia marcescens* Bacteriophage Selection Rationale

The *Serratia marcescens* bacteriophage fits most of the requirements mentioned above and has been successfully used in a variety of aqueous environments, including marine, as a tracer of hydraulic movement (Drury and Wheeler, 1982).

The host of the *Serratia marcescens* bacteriophage is not commonly found in the environment therefore it is less likely that the organism or its associated phage will be found in the environment. Cultivation and enumeration techniques are simple and materials required are commonly found in a basic microbiology laboratory. Adoptions of the Soft Agar Overlay Technique described by Adams (1959) and further improved by Skilton (1987) are commonly used. Production of large quantities of the bacteriophage have been described in a report on disinfection efficiency investigations in the ODA Final Research Report UK (Lloyd and Jones, 1992).

The *Serratia marcescens* bacteriophage demonstrated excellent survival in tap water compared to *Enterobacter cloacae* and *Escherichia coli* bacteriophages in a study carried out in Malawi (Watkins, 1987). Skilton and Wheeler (1988) reported good % recovery rates of 0.12-1.9% in groundwater (in an unconfined chalk limestone) and may survive for extensive lengths of time (Bitton and Gerba, 1984); phage were detected up to 12 months. *Serratia marcescens* bacteriophage was recovered from heavily polluted and turbid water with minimal difficulty as well as in river and seawater by Drury and Wheeler (1982). They carried out intensive experiments (bench-scale) to determine the $T_{90}$ in river, sewage,
and seawater. In sewage, it took 31-126.9 hours for 90\% inactivation; in seawater between 32.8-104.5 hours (in the dark). This indicates that although there is some loss of phage in saline environments, enough phage remains, sufficient to have suitable counts.

The NCIB 10654 *Serratia marcescens* bacteriophage strain was originally isolated from seawater by Dr John Watkins of Yorkshire Water, UK. It has a symmetrical polyhedral head, approximately 50 nm in diameter, a short tail and belongs to morphology group 3 (Tikhonenko, 1970). The host organism, *Serratia marcescens* is a gram-negative bacillus of the *Enterobacteriacae* group. The host strain for this bacteriophage is NCIB 10644. The host organism forms a red pigment, prodigiosin, when incubated at 25\°C, but not at 37\°C.

Carefully planned bench-scale experiments to evaluate and validate the application of *Serratia marcescens* bacteriophage as a tracer in waste stabilisation ponds were devised.

### 10.5 Phage Validation

In order to evaluate the feasibility of using *Serratia marcescens* bacteriophage (NCIB 10654) as a tracer of hydraulic retention time, evaluation tests were carried out. The main results obtained in these experiments are presented.

One of the main factors complicating the use of bacteriophage as tracer in stabilisation ponds arises from the uncertainty of the behaviour and survival of bacteriophage in such a dynamic and active ecological environment. Factors such as high pH, adsorption to suspended solids, sunlight and the presence of hydrogen sulphide in the pond liquors are likely to be major contributing factors to their survival. Due to the high sulphate loading experienced by these ponds, the ponds are normally anoxic even during the daylight hours and high levels of hydrogen sulphide are frequently measured in all effluents.

This study investigated whether the physico-chemical conditions expected in facultative and maturation ponds precluded the use of phage tracers and is published in Frederick and Lloyd (*in press a*).

### 10.6 Experimental Methods

Samples were collected at the sewage treatment works. A grab raw sewage sample was taken and samples from facultative pond 1.2 and maturation pond 2.2 were collected from the centre of the ponds using a water column sampler from a boat. The sampler, manufactured by International Lagoon Technology, was difficult to manage on the boat, the
The seal at the bottom was defective and had to be refitted with a caulking seal. Additionally one of the glued rope guides came apart from the main body further aggravating sample collection.

Basic operational parameters such as BOD, COD, pH, conductivity, sulphides, sulphates, DO, ammonia-nitrogen, suspended solids and faecal coliforms were analysed in the three samples: incoming sewage grab sample; facultative pond 1.2 and maturation pond 2.2.

Eleven different reactor systems were set up with 1 l volumes of raw sewage, facultative pond liquor, and maturation pond liquor in 1 l beakers. The sides and bottoms of the beakers were covered with black plastic in order to mimic light availability in ponds (Fig 10.4).

In order to assess the removal of the phage caused by adsorption to suspended particles; filtered (Whatman GF/D) and unfiltered samples of raw sewage, facultative pond and maturation pond liquids were placed in prepared beakers. The effect of hydrogen sulphide was investigated by comparing an aerated facultative pond sample with an unaerated facultative pond sample. The unaerated sample was placed in a ziploc plastic bag and all air removed through a small outlet attached (Fig 10.5). Aliquots were withdrawn by inserting a 5 ml volumetric pipette into the outlet. The contact of sample with air was kept to a bare minimum. The survival of the bacteriophage in high pH was observed by adjusting two samples of filtered maturation pond samples to pH 9.5 and placing one in the dark. As a control, a filtered maturation pond sample was also placed in the dark.

All experiments were placed outside (Fig 10.4) so that sunlight was available, except for the adjusted and non adjusted pH, filtered maturation pond samples which were kept in complete darkness thereby prohibiting photosynthesis. Because the experiment was outside and the month of June is in the rainy season, the experimental beakers were sheltered by a clear plastic sheathing approximately 4 m above.

A negative control test was performed on the raw sewage and pond samples. No plaques were formed therefore it was assumed that the phage of the host bacteria, *Serratia marcescens* was not present at detectable levels in the ponds ecosystem.

Samples were inoculated with 1.0 ml *Serratia marcescens* bacteriophage with a titre of $1.58 \times 10^{11}$/ml and stirred for 2 minutes. After 5 minutes a sample was taken from each
beaker and the ziploc bag. Detection and enumeration of the bacteriophage was done using an adaptation of the Soft Agar Overlay method (Adams, 1959). Media and growth procedures for the *Serratia marcescens* bacteriophage host bacteria are in Appendix IX.

All plates were incubated at 30-33°C; the red pigmentation characteristic of the host bacteria was evident. Another sample was collected and analysed after 60 minutes. Following that, samples were collected and analysed every day for four days after which samples were taken on the 6th, 7th and 13th day. Prior to sample collection, each reactor vessel was stirred for 2 minutes. Temperature and pH were measured each time a sample was withdrawn from the experimental reactor unit. On the 13th day, the experiment was terminated and the remaining samples were analysed for BOD, DO, pH, conductivity, sulphates, and ammonia-nitrogen.

Fig 10.4 Bench-scale experiment set-up to validate the feasibility of *Serratia marcescens* bacteriophage as hydraulic tracer.
Fig 10.5 Experiment 8 'sulphide effect' showing the airtight reactor vessel (at the end of experiment) and specially designed sample outlet in the left hand corner. In the background are beakers from some of the other experiments.

10.6.1 Results and Discussion

Results of bacteriological and physico-chemical analyses of samples prior to being prepared for phage experiment and upon completion of experiment (13th day) are shown in Tables 10.1 and 10.2:

Table 10.1 Bacteriological and physico-chemical quality of raw sewage and pond effluents prior to bench-scale department.

<table>
<thead>
<tr>
<th>SAMPLE</th>
<th>BOD uf</th>
<th>BOD f</th>
<th>COD uf</th>
<th>COD f</th>
<th>DO</th>
<th>FC BacT</th>
<th>pH</th>
<th>EC</th>
<th>SO₄</th>
<th>H₂S</th>
<th>NH₃</th>
<th>SS</th>
<th>VSS</th>
<th>FSS</th>
</tr>
</thead>
<tbody>
<tr>
<td>ISG</td>
<td>176.6</td>
<td>81.4</td>
<td>403.1</td>
<td>345.5</td>
<td>n/a</td>
<td>3.16E+06</td>
<td>7.55</td>
<td>13360</td>
<td>800.0</td>
<td>7.0</td>
<td>32.0</td>
<td>74.8</td>
<td>68.2</td>
<td>5.6</td>
</tr>
<tr>
<td>P 1.1</td>
<td>61.6</td>
<td>35.5</td>
<td>270.7</td>
<td>201.6</td>
<td>0.6</td>
<td>2.15E+05</td>
<td>8.02</td>
<td>14510</td>
<td>640.0</td>
<td>10.0</td>
<td>16.0</td>
<td>72.4</td>
<td>71.1</td>
<td>1.3</td>
</tr>
<tr>
<td>P 1.2</td>
<td>36.1</td>
<td>17.5</td>
<td>213.1</td>
<td>144.0</td>
<td>0.6</td>
<td>1.25E+05</td>
<td>8.01</td>
<td>14530</td>
<td>680.0</td>
<td>5.0</td>
<td>16.0</td>
<td>103.7</td>
<td>94.2</td>
<td>9.6</td>
</tr>
<tr>
<td>P 2.1</td>
<td>37.1</td>
<td>23.8</td>
<td>201.6</td>
<td>149.7</td>
<td>0.7</td>
<td>4.87E+04</td>
<td>8.09</td>
<td>14570</td>
<td>640.0</td>
<td>10.0</td>
<td>12.8</td>
<td>117.1</td>
<td>110.4</td>
<td>6.7</td>
</tr>
<tr>
<td>P 2.2</td>
<td>23.6</td>
<td>9.9</td>
<td>120.9</td>
<td>103.6</td>
<td>0.6</td>
<td>5.83E+03</td>
<td>8.17</td>
<td>15100</td>
<td>660.0</td>
<td>3.0</td>
<td>9.6</td>
<td>80.0</td>
<td>75.0</td>
<td>5.0</td>
</tr>
</tbody>
</table>

*All results in mg/l unless otherwise indicated.*
Table 10.2 Bacteriological and physico-chemical quality of remaining bench-scale experiments' effluents after termination.

Routine Analysis on Experiments Terminated 17 June 93

<table>
<thead>
<tr>
<th>Experiment</th>
<th>BOD uf</th>
<th>DO</th>
<th>FC BacT</th>
<th>pH</th>
<th>EC</th>
<th>SO₄</th>
<th>NH₄</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. ISG uf</td>
<td>11.1</td>
<td>8.7</td>
<td>1.00E+01</td>
<td>8.98</td>
<td>13970</td>
<td>960.0</td>
<td>3.2</td>
</tr>
<tr>
<td>2. ISG f</td>
<td>12.3</td>
<td>9.6</td>
<td>0.00E+00</td>
<td>9.10</td>
<td>14030</td>
<td></td>
<td></td>
</tr>
<tr>
<td>3. P1.2 uf</td>
<td>12.6</td>
<td>11.0</td>
<td>2.00E+01</td>
<td>9.27</td>
<td>16310</td>
<td>960.0</td>
<td>3.2</td>
</tr>
<tr>
<td>4. P1.2 f</td>
<td>12.3</td>
<td>9.8</td>
<td>5.00E+00</td>
<td>9.12</td>
<td>16060</td>
<td></td>
<td></td>
</tr>
<tr>
<td>5. P2.2 uf</td>
<td>11.5</td>
<td>8.8</td>
<td>0.00E+00</td>
<td>9.06</td>
<td>17810</td>
<td>1000.0</td>
<td>3.2</td>
</tr>
<tr>
<td>6. P2.2 f</td>
<td>12.7</td>
<td>10.5</td>
<td>0.00E+00</td>
<td>8.94</td>
<td>17340</td>
<td></td>
<td></td>
</tr>
<tr>
<td>7. P1.2 uf aerated</td>
<td>12.3</td>
<td>10.0</td>
<td>0.00E+00</td>
<td>9.10</td>
<td>17340</td>
<td>960.0</td>
<td>3.2</td>
</tr>
<tr>
<td>8. P1.2 uf un aerated</td>
<td>10.8</td>
<td>5.2</td>
<td>2.40E+03</td>
<td>9.48</td>
<td>13530</td>
<td>800.0</td>
<td>3.2</td>
</tr>
<tr>
<td>9. P2.2 f in dark</td>
<td>2.7</td>
<td>8.2</td>
<td>6.00E+00</td>
<td>8.71</td>
<td>15580</td>
<td>760.0</td>
<td>1.6</td>
</tr>
<tr>
<td>10. P2.2 f pH9.5 in dark</td>
<td>2.2</td>
<td>8.4</td>
<td>0.00E+00</td>
<td>8.85</td>
<td>15690</td>
<td></td>
<td></td>
</tr>
<tr>
<td>11. P2.2 f pH9.5 in light</td>
<td>10.2</td>
<td>9.0</td>
<td>2.00E+00</td>
<td>8.94</td>
<td>18100</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The data on bacteriophage concentrations, pH and temperature at the time of sampling are reported in Appendix X and are discussed in the following Sections 10.6.2 to 10.7.

10.6.2 Raw Sewage Experiments 1 and 2

Experiments 1 and 2 were designed to investigate and demonstrate the influence of sewage solids on adsorption and hence removal of the bacteriophage. The suspended solids in raw sewage in the present study were relatively low (average 150 mg/l) and at this level no significant differences in the phage counts were observed when comparing unfiltered with filtered raw sewage (Fig 10.6). Thus removal differences attributable to transport from liquid to the solids phase were not demonstrated and this is important from the point of view that sedimentation of the solids would not affect the use of this phage as a tracer.

Removal by adsorption and sedimentation is distinct from die-off and it is important to distinguish between transport of virus to sediment, where they are likely to survive for protracted periods and die-off in the wastewater column. It was assumed here that filtration by 2 Whatman GF/D filters removed sufficient sewage solids to allow a major part of the adsorption process to be prevented and die-off in the water column revealed.
Fig 10.6 The effect of raw sewage solids on the removal and survival of *S. marcescens* phage.

*Expt 1. ISG uf = inlet raw sewage grab sample, unfiltered.*

*Expt 2. ISG f = inlet raw sewage grab sample, filtered.*

Fig 10.6 demonstrates that die-off proceeds relatively rapidly in the supernatant sewage and in each experiment 2 distinct phases characterising die-off are discernible. The first phase is slower, with a loss of only 2 logs in 4 days. The second phase is rapid, with a reduction of 4-5 logs in 2 days. When this is compared with pH rise it can be seen that the die-off phases are inversely correlated with pH in each experiment.

The first, slower phase is associated with steadily rising pH from 7.5 to 9.0 during the first 4 days. However, when the pH 9.0 is exceeded the phage population collapses rapidly to undetectable levels in 2 days.

**10.6.3 Facultative Ponds Experiments 3 and 4**

In Fig 10.7 (experiments 3 and 4), although facultative pond effluent was used to study die-off, the pattern was similar to that seen with raw sewage on experiments 1 and 2. The most obvious difference presented in Fig 10.7 is that high pH (>9.0) was achieved
quicker, in one day for unfiltered and 2 days for the filtered facultative pond sample. This difference is associated with a higher starting pH of 8.0.

As a consequence the second phase of rapid phage decline also begins earlier, on day 2 (filtered) and day 3 (unfiltered). The associated effect in this rapid phase of decline is a 6 log reduction in 4 days for the filtered facultative sample, and a 6 log reduction in 3 days for the unfiltered sample. However the end point is the same, that is reduction or removal to undetectable levels in 6 days for all four experiments.

![Graph showing the effect of sewage solids on the removal and survival of S. marcescens phage.](image)

**Fig 10.7** The effect of sewage solids on the removal and survival of *S. marcescens* phage.

*Expt 3. P1.2 uf = facultative pond 1.2 sample, unfiltered.*

*Expt 4. P1.2 f = facultative pond 1.2 sample, filtered.*

### 10.6.4 Effect of Hydrogen Sulphide in Facultative Pond - Experiments 5 and 6

In experiments 5 and 6 the effect of hydrogen sulphide was investigated. High concentrations of hydrogen sulphide a characteristic of pond effluents due to the conversion of sulphate derived from saline intrusion. In experiment 5 an unfiltered
facultative pond sample was aerated to remove hydrogen sulphide and compared with an unaerated sample containing hydrogen sulphide. A clear difference is demonstrated in Fig 10.8 indicating that rather than having a toxic effect, the hydrogen sulphide provides a degree of protection for the phage, presumably due partially to the inhibition of photosynthesis.

![Graph showing the effect of hydrogen sulphide on phage removal and survival]

Fig 10.8 The effect of hydrogen sulphide on the removal and survival of *S. marcescens* phage.

*Expt 5. P1.2 uf = facultative pond sample, unfiltered & aerated.*

*Expt 6. P1.2 uf = facultative pond sample, unfiltered & unaerated.*

In experiment 5 there were two distinct phases of phage removal. The slower, first phase being in the initial 3 days where there was a 2 log removal, the second decline was very rapid as by the 4th day no phage was detected, a 5 log removal in 1 day. This dramatic decrease again coincided with high pH of 9-9.5.

It was observed that experiment 6, the unaerated sample, exhibited a similar rate of decline as experiment 5 up until the 4th day, when the log removal was only 1. However, the phage die-off rate after the 4th day continued at a slower rate than that of the aerated sample. Complete disappearance of the phage was delayed until the 7th day. It is notable
that the visual greening, by algal growth, was also delayed (by a day) in experiment 6 (compared with parallel experiment 5) due to the persistence of hydrogen sulphide in the unaerated sample. The significantly higher pH in the unaerated sample may be due to the fact that more of the ammonia remains in the unoxidised state (microbial nitrification cannot proceed in the presence of H₂S), and suggesting a lesser role for ammonia as phagicidal agent when compared with photooxidation effects.

10.6.5 Maturation Pond Experiments 7 and 8

The initial pH of raw sewage was typically 7.5 in the presence of 30 mg/l NH₃-N. At the outlet of the primary facultative pond, where pH has risen by 0.5 pH units to almost 8 (mean), the NH₃-N was reduced to less than 15 mg/l, whereas in the maturation ponds effluents the average was 11 mg/l.

It is thus clear that pH rise is mediated more by photosynthetic uptake of CO₂ leaving an excess of OH⁻, rather than to the amount of ammonia present. However additional factors in photosynthesis may have an important influence on die-off (Curtis, 1990) rather than the particularly high levels of H₂S associated with the Cayman ponds.

Fig 10.9 The effect of solids on the removal and survival of S. marcescens phage.

Expt 7. P2.2 uf = maturation pond 2.2 sample, unfiltered.
Expt 8. P2.2 f = maturation pond 2.2 sample, filtered.
In the experiments 7 and 8, the starting pH of the maturation pond samples was 8.1 and this rises above pH 9 in both experiments by the 3rd day. As a consequence, the removal of the phage was complete by the 4th day.

As shown in Fig 10.9, the pattern of slow decline in the first 3 days and then a rapid drop coinciding with the highest pH was again observed. As in experiments 1 and 2, no appreciable negative influence effect of solids was detectable.

10.6.6 pH and Photosynthesis in Maturation Pond - Experiments 9 and 10 and 11

In experiments 9, 10 and 11 (Fig 10.10) the influence of sunlight on survival in relation to photosynthesis and pH was investigated by means of pH adjustment and exclusion of light.

![Graph showing pH and survival of S. marcescens phage](image)

Fig 10.10 The effect of pH, sunlight and algal growth on the removal and survival of S. marcescens phage.

Expt 9. P2.2f = maturation pond 2.2 sample, filtered, kept in dark.

Expt 10. P2.2f = maturation pond 2.2 sample, filtered, pH adjusted to 9.50 with 5N NaOH, kept in dark.

Expt 11. P2.2f = maturation pond sample, filtered, pH adjusted to 9.50 with 5N NaOH, kept in normal daylight.
The initial severe adjustment to pH 9.5 might have been expected to immediately and dramatically reduce the phage populations if pH per se were the key factor. In fact, the phage population in the dark remained stable for 2 days (Expt 10), and in both darkened beakers (Expt 9 and 10) the phage populations survived at least 13 days at detectable levels, even though the pH (in Expt 10) remained just below pH 9 without further artificial adjustment. By contrast, in experiment 11, in the light, Serratia phage became undetectable at day 6, thus confirming the great importance of photosynthetic related mechanisms for phage death as shown by Curtis (1990) for faecal coliforms.

10.7 Conclusions - *Serratia marcescens* Bacteriophage as a Hydraulic Tracer

The overall aim of the investigation was to establish whether *Serratia marcescens* phage survived for protracted periods in the ponds and thus permit their use as retention time tracers and indicators for dispersion and short circuiting. From the validation experiments previously presented the following conclusions were made:

1. *Serratia* phage survival times were very short when pH rose above 9 and the importance of photooxidation in concert with high pH for phage die-off is in agreement with the findings of Curtis (1990) and Pearson *et al* (1987d) but for faecal coliform removal. The findings also agree with Funderburg *et al* (1978) who found a good correlation between poliovirus removal, high pH and chlorophyll a.

2. The experimental conditions in the 1 litre samples of pond sewage most closely resembled the top 20-30 cm of the lagoon system (except when the beakers were completely darkened). Since light penetration, photosynthetic activity and atmospheric exchange is confined to the upper zone of lagoons (unless overturn/mixing is substantial), it was considered logical to model these conditions in most of the experiments, because they are known to produce the most adverse environment for bacterial survival, and would therefore also be likely to provide the most stringent test conditions for phage survival.

3. The results of experiments 1-8 confirmed that high pH/photooxidation upper zone conditions are extremely hostile to phage survival. However, they are not the dominant conditions in the full-scale, deeper pond environment. The dark experiments and pH's near to 8 are more typical of the bulk of pond liquid. The average annual pH in the facultative pond outlets in Grand Cayman was <8, whilst the theoretical retention time
is 12-15 days. The average annual pH in the 2 maturation ponds outlets was 8.1 together with theoretical retention times between 4 and 6 days each.

4. The fact that detection of phage beyond 13 days was possible in experiments in the dark, in spite of high initial pH (Expt 10), suggests that photosynthetic activity not only plays a crucial role in rapid phage death, but also that *Serratia* phage will survive long enough in the bulk of pond liquor to make their use as tracers feasible. It is therefore concluded that since very high phage titres can be produced *in vitro*, tracing *in vivo* should be possible if pH remains below 9, but impractical with pH>9 due to photooxidative, pH-associated rapid die-off.

10.8 Evaluation of Retention Time and Short-Circuiting in WSP
Retention (residence) time is considered to be amongst the most important parameters which influence pond performance with respect to the removal and elimination of pathogenic and parasitic microorganisms. It follows that additional physical parameters, particularly wind speed and direction, bottom topography, dead zones, inlet and outlet arrangements, and baffles can all affect mixing. If any of these factors cause short-circuiting of flow, performance will be reduced.

Pond performance for pathogen removal is substantially dependent upon retention time. This means that if 1% of sewage enters and leaves a pond in less than 24 hours performance can hardly exceed 99% removal efficiency. Similarly if 10% of sewage short-circuits and leaves the pond within 24 hours then performance is unlikely to exceed 90% removal. It thus becomes important to assess the level of short-circuiting and it was proposed to do this by the use of the non-indigenous bacteriophage of *Serratia marcescens*.

In Section 10.6, it was demonstrated *in vitro* that *Serratia marcescens* bacteriophage survived for sufficient time in Cayman sewage to allow its use as a tracer of dispersion and flow. The results of that study indicated that the phage could survive for more than 13 days provided the pH was below 9.0 and without the influence of photosynthesis.

The main objective of this full-scale study was to calculate mean retention time and to determine the existence and extent of short-circuiting in addition to the effect of wind speed and direction on the flow of sewage through the lagoons.
10.9 Methodology

In order to evaluate retention time certain information concerning the physical properties of the pond chosen for the study were necessary. This included data on sludge depth and operating volumes.

10.9.1 Physical Properties of Primary Facultative Pond 1.1

The sewage treatment works as previously explained in Chapter 3, consist of 2 primary facultative ponds operating in parallel followed by 2 baffled maturation ponds in series. The nominal retention time of primary facultative pond 1.1 for the period of the study was 11.5 days while that of the complete system was calculated to be 18 days.

The sewage flow to the works for the study period averaged 2600 m³/day. The design dimensions of the facultative pond chosen for this study are as follows: width 60 m, length 160 m, and depth 1.8 m. The pond volume was calculated based on the pond’s average water depth.

10.9.2 Sludge Accumulation Pattern in Pond 1.1

The bottom topography of this pond was determined using the "white towel method" as described by Pearson et al (1987a), and Chapter 4. A total of 96 points were measured and plotted on Figs 7.7 and 7.8.

The maximum sludge depth measured was 145 cm. The average depth on the west side was 57.4 cm. The overall average sludge depth was 28.9 cm while the rate of accumulation in this pond 1.1 is estimated at 5 cm per year.

10.9.3 Collection of Meteorological Data

Windspeed data was recorded daily on site from an anemometer. Rainfall was measured daily on site. Wind direction data were obtained from the CI Civil Aviation Meteorological Station located approximately 2 km away.

10.9.4 Preliminary Investigation of Surface Dispersion Pattern

In order to have a preliminary idea of the surface dispersion of the phage inoculum that would follow, one hundred oranges were introduced at the inlet of the primary facultative pond 1.1. Although this would not be representative of the mixing behaviour throughout the pond column it was considered a useful indication of where the flow from the inlet pipes was likely to surface. In fact, oceanographers often resort to similar, simple methods.
to determine ocean currents and speeds. The results were useful in determining where and how much of the phage inoculum would enter the pond from each inlet pipe.

In about half a minute after injecting the oranges into the inlet chamber of pond 1.1, 96% of the oranges were seen to emerge from the western inlet. It was noted that at the eastern inlet the remaining 4% came out. This suggests that there is a blockage hindering the flow through the eastern inlet pipe. The approximate points of emergence were located in the pond using the grid system (Fig 10.11) and were identified as sample points A₁ and B₁. This information was used to select the sampling points in the pond.

Within the first 20-30 minutes all the oranges had drifted towards the western banks and spread over a distance of 20 m. The leading oranges were already 20% of the distance along the length of the pond after 24 hours but they had become trapped in the surface scum.

10.9.5 Introduction of Bacteriophage Inoculum to Pond
Ten litres of the *Serratia marcescens* phage (NCIB-10654) inoculum with a titre of $10^{10}$/ml was dosed into the inlet of facultative pond 1.1 simultaneously with a strong pulse of raw sewage.

10.9.6 Sampling Strategy
For sampling purposes, the facultative pond was divided into an imaginary grid. The grid split the pond lengthwise into two halves and then each half into 5 sections so that the pond comprised of 10 equal areas (see Fig 10.11).

Fig 10.11 Schematic of Pond 1.1 indicating sample points.
The temperature in raw sewage compared with the temperature profile throughout the pond effluent does not vary more than 3-4°C (Chapter 6). Although there is some temperature stratification, with 2-3°C differences observed between the surface and 0.70 m depth in the pond (Figs 6.14-6.17, pg 137), the effect is not vastly significant. From this study, the average temperature of the incoming sewage and the effluent of this particular pond is 29.8°C and 28.8°C respectively. Adequate mixing was expected therefore it was decided that subsurface samples would be appropriate.

A clean, plastic (polyethylene) sample bottle was used at each sample point. Bottles were cleaned with 35% chlorine solution and rinsed with sterile water. Samples were collected ‘in-pond’ from a boat using the 175 ml plastic bottles. Samples were collected from a depth of 10-20 cm. Additionally, the outlets of all ponds were sampled at selected intervals.

To ascertain that the *Serratia marcescens* bacteriophage was absent from the pond environment and would not compromise the results, a sample of the pond liquor was taken prior to inoculation with the phage and analysed.

Immediately before addition of the phage inoculum samples were taken (from the inlet works and the outlet of pond 1.1) at time zero (T₀). Immediately after the phage was introduced to the incoming sewage, approximately T₅ min, samples were collected at point A₁ and B₁. The sampling regime was at 3 hours, 6 hours, then every 24 hours until the 12th day when samples were taken every 48 hours up until the 22nd day.

**10.9.7 Laboratory Methods - Full-Scale Tracer Study**

Samples were collected at each point in duplicate and combined at the laboratory. The initial titrations (T₀ to T₃ hrs) were carried out using sample dilutions ranging 10⁻¹ and 10⁻², and subsequent samples were analysed without dilution (0.1-0.5 ml). Plates with clear, uncontaminated and countable plaques were accepted. For each sample point, generally 2-3 replicates were carried out. Detection and enumeration of the bacteriophage was done using an adaptation of the Soft Agar Overlay method (Adams, 1959). Information on incubation history (temperatures, date and time); sample dilutions and raw counts were recorded.

The electrical conductivity, temperature and pH of each sample was measured in the laboratory and recorded (see Appendix XI). It was not possible to take temperature
readings immediately after collection due to operational difficulties with portable equipment therefore that data is not included in the results of the study shown in Appendix VIII. The pH and electrical conductivity of the samples were analysed using the appropriate calibrated electrodes and meters.

10.10 Results of Full-Scale Tracer Study
The phage results in Table 10.3 show that more than 30% of the surviving inoculum transited the dosed pond in less than 6 hours.

Table 10.3 Serratia marcescens bacteriophage pfu/ml recovered from the outlet of pond 1.1 6-28 Sep94.

<table>
<thead>
<tr>
<th>TIME</th>
<th>Pond 1.1 Outlet</th>
</tr>
</thead>
<tbody>
<tr>
<td>T₀</td>
<td>0.00E+00</td>
</tr>
<tr>
<td>T₁₂₄₆₇</td>
<td>1.98E+03</td>
</tr>
<tr>
<td>T₂₄₆₇</td>
<td>1.55E+03</td>
</tr>
<tr>
<td>T₄₆₇</td>
<td>4.58E+02</td>
</tr>
<tr>
<td>T₇</td>
<td>4.48E+02</td>
</tr>
<tr>
<td>T₉</td>
<td>2.66E+02</td>
</tr>
<tr>
<td>T₁₀</td>
<td>3.36E+02</td>
</tr>
<tr>
<td>T₁₁</td>
<td>1.50E+02</td>
</tr>
<tr>
<td>T₁₂</td>
<td>9.40E+01</td>
</tr>
<tr>
<td>T₁₃</td>
<td>1.11E+02</td>
</tr>
</tbody>
</table>

The counts of phage recovered from the outlet of the facultative pond 1.1 (Table 10.3) were compared with the dose of phage injected at the inlet. The 10 l of phage suspension introduced at T₀ contained a total of 10¹⁴ phage particles. The 3 hour outlet sample contained 1.98 x 10³ ml⁻¹. It is estimated that >50 m³ of primary effluent leave the pond every hour. This converts to approximately 1 x 10¹¹ phage leaving the pond during the hour after the first sample and comprise 0.1 - 0.2% of the original inoculum. It is thus likely that as much as 1% or more of the total inoculum has passed out of pond 1.1 in the first 12 hours. As a consequence it is not expected that the facultative pond can significantly exceed 99% faecal coliform removal efficiency.
In order to determine the mean retention time $\bar{t}$, and to describe the hydraulic performance of the pond using the dispersion index, $d$, in the method of Levenspiel (1962) and the following equations were used:

$$\bar{t} = \frac{\sum t_i C_i}{\sum C_i}$$  \hspace{1cm} \text{Eq 10.4}$$

where $t_i$ = time elapsed from injection of tracer
$C_i$ = concentration of tracer in sample at $t_i$

The dispersion index, $d$, which is used to describe types of flow dispersion may be calculated from the adimensional variance which is defined by:

$$\sigma_i^2 = \frac{\sum t_i^2 C_i}{\sum C_i} - \left[ \frac{\sum t_i C_i}{\sum C_i} \right]^2$$  \hspace{1cm} \text{Eq 10.5}$$

$$\sigma_i^2 = \sigma_i^2 - 2d^2(1 - \exp(-1/d))$$  \hspace{1cm} \text{Eq 10.6}$$

where $\sigma_i^2$ = adimensional variance
$\sigma_i^2$ = variance of the $C_i$ vs $t_i$ curve ($t_i^2$)
$d$ = adimensional dispersion index

The deviation from plug flow (dispersion index = 0) of the value of the variance calculated based on the phage tracer data was high (2.7) indicating that the flow through was nearer to that of a completely mixed system (dispersion index = $\infty$). The data from Table 10.3 is plotted in Fig 10.12. In this graph it is clear that a significant amount of tracer exited the pond in the first 3 hours.

Fig 10.12 *Serratia marcescens* bacteriophage in effluent from outlet of pond 1.1 throughout the tracer study period.
Analysis of the area under the curve of Fig 10.12 demonstrated that 36% of the phage surviving the transit through pond 1.1 pass through the outlet in 20 hours.

The data in Table 10.3 are also plotted in a dimensionless graph (Fig 10.13) to more clearly illustrate the short-circuiting occurring in the first 24 hours and the consequent poor mixing.

![Dimensionless tracer concentration Ce/Co as a function of dimensionless time t/T. (using 24hr data points, 12 days)](image)

where $Ce = \text{tracer measured at time } t; Co = \text{tracer concentration if completely mixed in system}; T = \text{theoretical retention time}.$

Fig 10.13 Dimensionless graph of *Serratia marcescens* bacteriophage measured at outlet of facultative pond 1.1.

In Fig 10.13, short-circuiting is illustrated as significant concentrations of tracer show up at $t/T$ of less than 0.3-0.4. From the variance of mean $t$, the dispersion number or index, $d = 2.743$. This fairly high value indicates that a large fraction of the tracer (and therefore flow) exit the pond in less time than the mean retention time of 3.79 days.

The movements of the phage through pond 1.1, studied by the more detailed sampling of the 10 sections in the pond, are illustrated in Figs 10.14-10.20 and show how phage became dispersed and declined. The contour graphs are drawn using a graphic programme, Surfer Version 6.
Wind: 40 degrees from N at 2.1 m/sec

Phage distribution after T3 hours

Phage distribution after T6 hours

Fig 10.14 *Serratia marcescens* bacteriophage distribution in facultative pond 1.1 after 3 hours and after 6 hours.
Wind: 70 degrees from N at 3.1 m/sec

Phage distribution after T1 day

Wind: 110 degrees from N at 2.1 m/sec

Phage distribution after T2 days

Fig 10.15 *Serratia marcescens* bacteriophage distribution in facultative pond 1.1 after 1 day and after 2 days.
Wind: 110 degrees from N at 2.6 m/sec

Fig 10.16 *Serratia marcescens* bacteriophage distribution in facultative pond 1.1 after 3 days and after 9 days.
These figures demonstrate that the windward (eastern) half of the lagoon has only a minor fraction of the flow, while in the western half the high wave of phage moves down to the outlet in the first 24 hours. This confirms the outlet counts presented in Table 10.3 and Fig 10.12.

10.11 Discussion
These findings were confirmed in more detail by Fares and Lloyd (in press) by applying the Fares (1993) model to the Cayman physical data of pond dimensions with windspeed and wind direction data for selected days in September 1994 when the field experiment was taking place. The Fares model is based on the theory of shallow water equations developed for predicting circulation patterns in lakes. The model demonstrated that with wind from the northeast, the flow is split diagonally into two cells rotating in opposite directions (Figs 10.17 and 10.18). Fig 10.19 is a windrose showing the predominant direction and windspeeds during the full-scale experiment. This data is incorporated in the model applied by Fares.

Many authors refer to the ability of wind to mix and cause overturn in ponds but few consider the influence of wind in short-circuiting. Exceptionally it is recommended that lagoon layout should be planned so that “the direction of the prevailing wind is never along the line of flow thus short-circuiting sewage from inlet to outlet or retarding normal flow” (AID 1970).

Ellis (1983) asserts that “wind action always improves the efficiency of ponds, principally by inducing vertical mixing which not only destroys stratification but greatly increases the rate of oxygen transfer”, hence (he suggests) wind must always be allowed free access. By contrast Mara et al (1992) point out that “to minimise hydraulic short-circuiting, the inlet should be located such that the wastewater flows in the pond against the wind”. Fares and Lloyd (1995) go further and apply a numerical model based on shallow water equations (Fares, 1994) to confirm the existence of short-circuiting and emphasise the significant wind effects on the hydraulic behaviour of WSPs in Grand Cayman.

The Fares model also confirmed that for a windspeed of 2.6 m sec\(^{-1}\), the residence time for around 40% of the inflow would be about 6.6 hours and even less at higher windspeeds. Additional figures are presented which show the effect of different wind directions on the sewage flux within the pond (Figs 10.20-10.23).
Fig 10.17 Predicted volume flux variations in facultative pond 1.1 using Fares' numerical model with wind 50° from north and speed of 2.6 m/sec during phage experiment in Sep94.
Fig 10.18 Simulated volume flux variations contours in facultative pond 1.1 using Fares numerical model with wind 50° from north and speed 2.6 m/sec during phage experiment in Sep94.
Fig 10.19 Daily windspeed and wind direction at the sewage treatment works during the full-scale phage tracer experiment 1-30 Sep94.
Fig 10.20 Predicted volume flux variations in facultative pond 1.1 using Fares’ numerical model with wind 90° from north and at 3.0 m/sec.
Fig. 10.21 Predicted volume flux contours in facultative pond 1.1 using Fares' numerical model with wind 90° from north and at 3.0 m/sec.
Fig 10.22 Predicted volume flux variations in facultative pond 1.1 using Fares' numerical model with wind 180° from north and at 3.0 m/sec.
Fig 10.23. Predicted volume flux contours in facultative pond 1.1 using Fares' numerical model with wind 180° from north and at 3.0 m/sec.
McDonald and Ernst (1986) studied 2 Australian pond systems in order to compare calculated and actual retention times. Using rhodamine dye as a tracer they showed calculated retention times of 34 and 16 days to be reduced to 38 hours and 12 hours respectively. They concluded that short-circuiting was due to design deficiencies and thermal stratification. In the case of the Cayman ponds the temperature difference between the top and bottom layer is on average no more than 2-3°C. Therefore stratification is not likely to be significant when compared with even low windspeeds.

Other studies have indicated that dependence on theoretical retention time to predict effluent quality from WSPs is unreliable. Gaillard and Crawford (1964) conducted fluorescein tracer tests on three maturation ponds (rectangular, length double the width) used to treat humus tank effluent and settled sewage in South Africa. They found that although the theoretical time was a week, the dye passed through in a few hours indicating short-circuiting. Apparently the dye did not reach large portions of the ponds which were therefore being under-utilised.

Uhlmann (1980) points out that strong wind action does not produce complete mixing of sewage lagoons and that they are relatively resistant to mixing at temperatures above 15°C when wind stress is moderate. He states that the inflowing sewage will tend to move into the water layer with the same density (temperature) and a short-circuit flow, by-passing a large portion of the ponds’ volume may result.

Uhlmann (1980) goes on to say that “in tropical lowlands at low altitudes, with high humidity and a small diurnal amplitude in air temperature, nocturnal overturn is (unusual) an exception. Accordingly, the hydraulic short-circuiting is the normal flow-through pattern. Thus the effluent quality is expected to deviate substantially from the (design) expectations, because neither the design equation for complete mixing nor the equation for piston flow apply to this case.”

10.12 Maturation Ponds
The phage leaving the facultative pond 1.1 was further tracked through maturation ponds 2.1 and 2.2 and in the recirculated effluent through facultative pond 1.2 (Figs 10.24-10.27). This revealed that a peak of phage left maturation pond 2.1 on the 4th day of the experiment suggesting that the most important fraction transits and leaves this pond in only 2 days (comparing Fig 10.24 and 10.25).
The retention time in pond 2.2 is much less since the first peak also exits pond 2.1 on the 4th day, indicating major short-circuiting in less than 24 hours.

It is also interesting to note that a second portion of the peak leaves both maturation ponds within 24 hours of each other (day 7) which further emphasises the severity of short-circuiting. This second peak is also seen exiting pond 1.1 on the 4th day.

The persistence of the phage is seen not only in pond 1.1 where it continues to decline as it mixes and dies but is still recoverable after 22 days, but also in the recirculated effluent where it is detected the second time round at the outlet of pond 1.2.

The results of the tracer experiment clearly indicate that there is extensive short-circuiting occurring in the ponds. Controlling the bacterial quality of the effluent is compromised due to the short-circuiting. A plug-flow hydraulic regime would improve the retention time and thus improve the removal efficiency of the system. However the practicality of achieving this may be difficult in the CI operational system.

10.12 Conclusions

The main purpose of this study was to establish the existence and extent of short-circuiting and actual retention time in primary facultative ponds in the Cayman Islands. This study also confirmed that Serratia marcescens bacteriophage is a suitable tracer of hydraulic retention times in facultative ponds.

- It was demonstrated that short-circuiting is associated with a prevailing easterly/northeasterly wind.

- The level of short-circuiting compared with the design retention time of 11.5 days shows 90% of the surviving inoculum passing through the pond in less than 6 hours therefore the removal efficiency is severely compromised.

- Reduced faecal coliform removal in the Cayman ponds may be associated with short-circuiting and increased flow whereas the BOD removal deterioration may be associated with high saline sewage.

Finney and Middlebrooks (1980) evaluated an empirical design equation proposed by Larsen in 1974. The design took into account; windspeed; solar radiation; relative
humidity; air temperature; lagoon temperature; air temperature; influent flow rate; influent BOD; effluent BOD; and pond area. Finney and Middlebrooks (1980) reported that according to Larsen, his design equation could be applied to any geographical area because it is in a dimensionless form. They did not however discuss the relevance of wind direction.

Arthur (1981) in a 12 month intensive study of pilot-scale waste stabilisation ponds in Nigeria found correlations between bacterial removals and climatic parameters such as wind speed, rainfall, relative humidity, evaporation and sunshine not to be highly significant. He did find significance between faecal coliform removal rates and temperature to be significant at 5%. This study included a comparison between completely stirred and unstirred ponds, the results indicated a slight BOD removal advantage in the mixed ponds over the unmixed ones. Temperature was found to be the most significant controlling factor in the BOD removal. The emphasis placed on windspeed in this study was limited, however it was implied that in full-scale ponds a good wind and thermal mixing is beneficial to treatment. No reference was made to what implications direction may have if any on the performance of waste stabilisation ponds.

Recommendations have been made that aerobic (maturation) ponds be less than 10 acres in size to minimise wind-driven short-circuiting (Metcalf and Eddy, 1972).

So, it is clear that these authors were aware of the relevance of wind direction and speed effects on large bodies of water as reported by Fares (1993, 1994) and Thackston et al (1987). It is probable that in earlier studies researchers have not had access to the shallow water equations adapted and developed by Fares. Since the means to model the hydraulics numerically were not available these potentially important factors were overlooked until now.
CHAPTER 11

11.0 FINAL DISCUSSION ON WSP SYSTEM PERFORMANCE, RESIDENCE TIME, CONCLUSIONS & RECOMMENDATIONS

Whereas it was considered at the beginning of this research that it would be academically worthwhile investigating predator/prey ecology, in the light of the very major practical problems of saline intrusion revealed by the monitoring programme it was decided to re-define the objectives to undertake a more applied study of faecal coliform indicator die-off, removal and hydraulic retention time.

In view of the hydraulic overload, the limited information available locally at the time of the design and the various design models that were available it is a tribute to the designers of the treatment system and the robustness of the system that the ponds have been able to perform with reasonable success. A faecal coliform removal efficiency of >3 logs was achieved for 4 out of the first 7 years of the life of the system, compared with the 4-5 logs predicted at the design stage. The extent of the problem of saline groundwater intrusion and hence hydraulic overload could not have been foreseen yet for 2 years (1989-90) the performance held above 4 log removal.

11.1 General Performance Evaluation of WSP System

- Salinity, although an important factor in the performance of the WSP system, was primarily indicative of excessive flow. The monitoring study has highlighted the importance of both hydraulic overload and the attendant salinity which derived from groundwater.

- R² correlations between faecal coliform removal and BOD were not high. This indicates that the BOD does not increase as there is more flow in fact the converse is true for the Cayman system. This is because as the groundwater is entering the system it is diluting what is ‘true’ sewage thus reducing the level of BOD. One would expect that the faecal coliform data would reveal a similar trend, however as seen in Chapter 5, there was a protracted period where the faecal coliforms were increasing as the salinity was increasing and the BOD decreasing.
• One major achievement of this thesis is to have tracked the decline and recovery of performance which could only have been done with a study over a protracted period (>5 years). The long term analysis of performance has clearly shown that the hydraulic overload due to saline groundwater intrusion into the sewers has further 'complicated' the already complex mechanisms that are characteristic of WSP systems.

• The overall effect of salinity through microbial conversion to hydrogen sulphide is relatively ineffective in reducing performance in the facultative pond as compared with a major reduction in removal caused by hydrogen sulphide carry over to the maturation ponds (data from Sept-Oct88). This coincides with pH in the maturation ponds falling below 9 as the free hydrogen sulphide starts to inhibit algal growth. Faecal coliform removal in maturation pond 2.1 never achieved >96% after Aug88.

• Sulphide toxicity to algal population was identified and noted that algae are the most vulnerable component of a WSP treatment process to hydrogen sulphide. pH is important because it controls the relative amount of hydrogen sulphide and ammonia-nitrogen present in different toxic forms. The Cayman ponds are organically underloaded and are very rich in sulphates. This author submits that a maximum allowed concentration of sulphate in the raw sewage of 300 mg/l for WSP systems in the Cayman Islands.

• The major components that are characteristics of domestic sewage in the CI is weak due to the influence of tourism, and groundwater infiltration. This information will be useful for further expansion and design of sewage treatment works. The monitoring programme continues to generate data which may be used to develop design standards in saline WSPs in other countries. Evaluation of the feasibility of using saline WSP for domestic wastewater treatment is an area for further research.

The monitoring programme was successful in investigating the effects of elevated salinity levels on the WSP and continues to serve the WAC with the collection of routine data.
The functional ecology study, further developed, could aid in defining the conditions and effectiveness of treatment ponds treating saline domestic wastewater, in providing suitable environments for the removal of pathogenic bacterial populations through protozoa and rotifers predation. Additionally, more practical information would be available for engineers and scientists to decide optimum treatment for saline wastewater. Many touristic coastal towns in semi-arid and arid areas are turning to seawater for toilet-flushing due to increased water production and treatment costs, or face depletion of fresh water resources. Thus the need for effective wastewater biological treatment methods for saline sewage will become more important.

11.2 Virus Removal and Indicators

The efficiency of virus removal in the wastewater treatment system is of concern because of the potential use of final pond effluent as irrigation water, after the salinity is reduced, on golf courses in the major tourist areas of Grand Cayman.

In Feachem, et al (1983) the survival time of enteric viruses in soil is given as <100 but usually < 20 days. Rose and Gerba (1991) report that Arizona is the only state in the United States which has implemented standards for enteric viruses in reclaimed water used for unrestricted irrigation, a standard of 1 enteric virus/40 l. They reported that oxidation lagoons were able to produce effluents that averaged 2 pfu/40 l. The quantitative detection of pathogenic enteroviruses is expensive, time-consuming and requires specialised cell culture techniques therefore limiting the monitoring of such viruses to specialised laboratories.

There are on-going studies which aim to develop efficient procedures such as polymerase chain reaction (PCR) for the detection of viruses (Dizer et al, 1993), but these are unlikely, to be widely available in the near future.

In order for the virus model function of bacteriophage to be acceptable it is essential that their resistance to treatment processes be similar or slightly greater than that of relevant pathogens. In a review paper by the IAWPRC (1991) study group on health-related microbiology it was noted that viruses are a heterogeneous group and their resistance or removal cannot be universally defined.
The *Serratia marcescens* bacteriophage has been used as an index of the survival of human enteric viruses in conventional sewage treatment plants such as activated sludge (Carstens *et al.*, 1965). This phage has also been used as a measure of viral removal efficiency of disinfection methods for potable water (Lloyd and Jones, 1992). It is suggested here that they may also be used as models for virus removal in facultative and maturation waste stabilisation ponds since they are similar in size and composition to many enteric viruses. Virus removal by other wastewater treatment processes has been reviewed extensively by Leong (1983); Bitton (1987); and Lewis and Metcalf (1988).

Scarpino (1978) reviewed the topic of virus indicators and concluded that phage may serve as indicators of pollution in some environments but not in others. The premise has been adopted here that the removal and survival characteristics should be investigated and understood at different stages of sewage lagoons to assess whether they may be used as indicators of the removal of pathogenic viruses and/or tracers of flow and dispersion. Few research studies have effectively partitioned the effectiveness of various mechanisms responsible for die-off and removal of pathogens or indicators.

Most early work on conventional sewage treatment suggested that primary settling of raw sewage, typically 4-8 hours, produced little or no reductions in the levels of indigenous viruses (Bloom *et al.*, 1959; Mack *et al.*, 1962; England *et al.*, 1967; and Berg, 1973). Berg suggested that most of the viruses in raw sewage are solids associated and that longer settling times of >12 hours should result in considerable virus removal. More recently Funderburg *et al.* (1978) and Oragui *et al.* (1987) suggested that in ponds (lagoons), sedimentation of solids associated virus is an important mechanism for removal. High pH has also been linked with high virus removal and the effectiveness of high pH coagulants (such as lime) has been demonstrated by Taylor *et al.* (1991) but the role of pH *per se* or in association with photosynthetic effects remains a conundrum. However, for phage, the evidence produced in Chapter 10 indicates that it is not experimental pH *per se* but the associated photosynthetic activities that affect survival.

### 11.3 The Importance of Wind on Hydraulic Performance and Design

It has been demonstrated that *Serratia marcescens* bacteriophage is useful as a tracer under the anoxic conditions of the Cayman WSPs, the same may not necessarily be true in aerobic ponds.
Marais (1974) observed an inverse relationship between coliform densities and windspeed. He concluded that this was due to de-stratification. This is further amplified in a study carried out by Moeller and Calkins (1980). They concluded that windy conditions improved mixing and brought more bacteria to the surface causing greater die-off due to UV-B exposure. In the present study, however windspeed and direction were considered to be driving the flow so as to cause short-circuiting and reduced microbial removal.

Although in the Cayman Islands, the prevailing winds are easterly for a large part of the year, northeasterly winds are also common (CICA Meteorological Station, 1995). Wind in the northeasterly direction were shown to aggravate the already critical hydraulic overload. Sludge accumulation was also shown to be very uneven, probably due to a combination of factors, for example, wind direction and inlet arrangement.

From the literature review, it is clear that traditional designs methods and models for WSP systems do not attempt to include the hydraulic behaviour of the pond systems. Factors, such as the location of baffles, inlet and outlets, wind direction and speed, that logically affect the hydrodynamic behaviour of the system are not addressed by the design tools currently available to engineers. To date the mathematical design procedures have not been successful in adequately describing the complex hydro- and biodynamic behaviour of ponds.

Some attempt has been made with computational fluid dynamic (CFD) modelling (Wood et al, in press). However, this tool completely ignores the influence of wind on the hydraulic behaviour in ponds and the effects of pond depth were not included. The authors did note some of the inadequacies and pointed out that their objective was to highlight the potential usefulness of CFD modelling. Other researchers (Preul and Wagner, 1987) proposed a complex analytical model to predict removal efficiency for BOD. This model attempts to incorporate sludge mass and temperatures into the calculations. However, the hydraulic flow pattern in ponds used, is an adaptation of that assumed by Ferrara and Harleman (1981). This model is considered to consist of 3 reactors one of which is a plug-flow reactor and the others completely mixed flow reactors. The study did not attempt to include or address the effects of wind on mixing in the ponds.
The Fares (1993) and Fares and Zaki (1994) numerical model is potentially extremely useful because it can be used for simulation of different arrangements and configurations and very importantly includes the effects of wind direction and wind speed! Further validation is recommended with tracer studies to obtain experimental and numerical comparisons which could assist in further developing the model for a wide range of pond configurations.

Some authorities are reluctant to accept WSP systems because it is practically impossible to predict how modifications to a system will improve performance. With a well-tested predictive tool available to pond designers and operators more confidence could be placed in expansion or rehabilitation of existing systems. The ‘acceptability’ of pond systems for wastewater treatment instead of conventional methods would certainly increase.

11.4 Conclusions and Recommendations

1. The tracer study results (and Fares’ numerical model confirms this) indicate that the strong pulses of sewage that enter the facultative ponds and the effect of wind directions from the E, ENE, NE at a speed of ≥2m/sec will together promote serious short-circuiting. It is recommended that the inlet structures in the future should be designed so as to minimise the jet velocity of the incoming sewage especially in systems that are not gravity fed but must be pumped due to gradient of the local topography. The existing structure in CI may be improved by the installation of baffles adjacent to the inlets of the facultative ponds. This would seek to remove the energy from the injected sewage thereby dissipating its velocity and consequently reduce the level of short-circuiting. Short-circuiting is aggravated by hydraulic overloading and this has led to poorer treatment efficiency than expected from the initial design.

2. It is recommended that recirculation of the effluent from the final maturation pond to the facultative ponds be stopped, except when septage is being emptied into the wet-well. Recirculation increases the hydraulic load by an estimated 20% and with the identified short-circuiting that occurs, retention time in the system is further reduced. Furthermore as effluent from the final maturation pond is withdrawn from almost 1 m below the top water surface level, it is rare that oxygen will be present in sufficient concentrations to improve the performance of the facultative ponds. This is clearly shown in the discussion on dissolved oxygen profiles in the ponds in Chapter 6. The other factor to be considered is that the microbiological quality is
poorer (higher faecal coliforms) the deeper one goes down into the pond. This factor is further emphasised as the windows in the maturation ponds' baffles are located in the 0.50-1.20 m zone (Fig 3.10, pg 54) below the top water surface level. As the incoming sewage is considered weak in terms of BOD (113 mg/l) it is not necessary to recirculate in order to reduce pond loading as suggested by (Benefield and Randall, 1980).

3. It is recommended that the draw-off levels for effluent from maturation pond 2.2 be raised above the present level of 1 m below the top water level. This zone of the pond is generally anaerobic or anoxic, with levels of oxygen rarely >0.5 mg/l (see Chapter 6, discussion on DO profiles). A take-off configuration with a scum guard to prevent carry over of floating scum from one pond to the other would be an improvement. For maturation ponds the draw-off level should be that which gives the best possible microbiological quality. The recommendation from Mara et al (1992a) is 0.05 m from the top water level, it is noted that at around this depth and above, oxygen is generally present during the day in the Cayman lagoons.

4. It is recommended that in the facultative ponds the effluent be withdrawn just below the maximum depth of the algal band and photosynthetic bacteria so as to minimise the quantity of algae and BOD leaving the pond (Mara et al, 1992a). In the CI pond system, this level is very close to the surface due to anoxic conditions that persist in ponds and the associated low levels of algae present (Chapter 9). The effluent draw-off point should be such that the quality is the best in terms of faecal coliform indicator bacteria and BOD removed. The effluent draw-off level recommended for the facultative ponds is 0.30-0.60 m. If the scum guard has variable height adjustment possible, optimal take-off at whatever level the ponds are being operated will be possible. This configuration would introduce more flexibility and operational control in managing the lagoon treatment system.

5. A proper flow measuring device, eg. a Doppler flow meter similar to that used for the inflow, is recommended for the final maturation effluent. This will permit the rate of evaporation to be calculated, in addition to the actual volume of effluent being discharged into the disposal wells.

6. As the design flow for 1996 (2813 m³/day) has been reached and will most likely be surpassed in 1995, it is recommended that the WAC take the next step in upgrading the treatment works to accommodate increased flow. If the system is to continue
producing effluent of acceptable and reasonable quality, which must be disposed of into the environment (disposal wells), then expansion will have to be considered in the very near future.

7. If additional WSP treatment systems are to be built in the Cayman islands, data and methods described in this thesis are applicable and should be utilised. It is recommended that the orientation of the system with respect to the prevailing easterly winds on Grand Cayman be considered a serious factor at the design stage. This research indicates that the hydraulic regime in terms of retention time, thus performance, as predicted with the Fares numerical model would be at its' optimum if ponds are orientated with their outlets towards the east instead of towards the south.

8. It is recommended that the WAC continue to play a key role in the development of the Fares' numerical design model by collaborating with the Centre for Environmental Health Engineering at the University of Surrey. This will not require the WAC to spend considerable sums of money, it would entail the continued collection of basic meteorological data and faecal coliform bacteria removal efficiency monitoring as is presently being done. The benefits to the WAC would be:
   a) the opportunity for other young Caymanians to carry out applied research towards a Masters or PhD degree and also as a project for Professional Engineering qualifications;
   b) the use of the model for design and prediction of pond system effluent quality;
   c) international recognition as a major participating centre for the improvement of low-cost wastewater treatment technology.

9. It is recommended that every effort continue to be made to improve the situation by reducing the intrusion of saline groundwater into the WSP system. In addition to repairs presently being carried out, diligent monitoring is continuing to ensure that when the intrusion has been reduced to the lowest possible level, that the acceptable level of salinity (2000-5000 μS/cm) is not exceeded. Additionally, the WAC and the responsible Ministry may wish to consider tightening the WA Regulations with respect to saline discharges and any other discharges (not normally found in domestic sewage in high concentrations) to the sewers which will be detrimental to the wastewater treatment method. Not only has the high flow and salinity affected the performance of the treatment system, the impact has also been felt by the WAC from the corrosive effect that the accompanying chlorides and sulphates/sulphides
have on the entire sewer system, including control panels, pumping stations, and manholes. The Authority has had considerable expense to effect repairs and rehabilitation of the very young sewer system. With careful routine monitoring of the salinity in the system, early detection of problems will be possible.

10. If the WAC is able to achieve a reduction in salinity to <5000 µS/cm, then it is recommended that a joint project with the Department of Agriculture be pursued. This may be done with a view to developing reuse possibilities (using effluent blended with fresh water) with saline tolerant crops and landscape plants. Discussions have already been held on an informal basis with the Department. Considerable interest was indicated in such a joint venture that may result in cost-effective and improved productivity benefits to local farmers. It is noted that pisciculture may not be possible as the effluent is unsuitable due to the high level of ammonia present which is toxic to fish.

11. The WAC may consider improving the final effluent through the use of high-rate algal ponds which are very shallow ponds, about 0.30-0.45 m deep. The final effluent from such ponds is generally of a better microbiological quality than that from traditionally designed maturation ponds, additionally there is the added potential attraction of harvesting aquatic macrophytes (duckweed) which may then be used for fertiliser or chicken feed.

12. The method of using the *Serratia marcescens* bacteriophage as a tracer of hydraulic retention time and dispersion within the lagoons is a useful and inexpensive way in which to periodically evaluate the WSP in the Cayman Islands. As a result of this research study, the expertise and material are now available locally to utilise this method when the need arises.

13. Additionally this method of tracing the movement of pollutants in aqueous media is applicable to other environments in the Cayman Islands such as groundwater and also in marine waters. Therefore the WAC may consider utilising this tracer tool and method in the management and monitoring of the groundwater resources of these islands. The *Serratia marcescens* bacteriophage or other suitable bacteriophages can also be used to assess the risk of the movement of pollutants in aqueous environments such as landfill leachate, wetlands and coastal waters.
11.5 Recommendations for Further Research

The wastewater treatment lagoon technology is low-cost and continues to provide communities around the world with an effluent that is generally far superior in terms of pathogen removal than that produced by conventional, high-energy consuming technology. For many of the world's arid and semiarid regions the use of reclaimed water for agricultural irrigation could mean the difference between drought and a successful harvest. WSP systems, even with the intricate complexities of this 'natural' wastewater purification method, are still able to produce the most suitable effluent for agricultural irrigation at the least cost compared with conventional systems. For countries concerned with management of their water resources, agricultural irrigation with the final WSP effluent is a very attractive alternative for disposal, provided that there are few harmful compounds or elements in the wastewater being treated.

Whether or not the final effluent is to be reused or discharged to the environment, the responsible agency will be concerned to demonstrate that there is the least risk possible to the human population and to the environment. Therefore the final effluent quality and the prediction of that quality in operational systems will be a useful and cost-effective tool.

Consequently, there are excellent possibilities for the application of the Fares’ numerical model as a design tool for wastewater treatment lagoons. Although Fares (1993) developed this predictive model for the Red Sea, the potential for its adaptation to shallow basins has been demonstrated with the preliminary valuable comparison of results obtained from the *Serratia marcescens* bacteriophage tracer study.

Its development will entail much detailed research for wind direction, windspeed, and pond orientation, in addition to other meteorological data over long and consecutive periods of time. Research studies using experimental and full-scale systems with variations on layout, length/width ratios, inlet/outlet configurations, and comparisons with full-scale operating systems will assist in the development and use of the model as an effluent quality prediction and design tool.

The further development of the model may provide a method that includes one of the most influential factors (i.e. wind direction and windspeed) on the mixing regime in ponds whose inclusion in design models/equations has rarely been attempted due to its inherent complexity. Of course, the debate will continue on whether a 'plug flow' or a 'completely mixed' system is the most efficient hydraulic regime for wastewater
treatment ponds. Nevertheless, the world is far from ideal, and no matter what the designer intends, practicalities will continue to dictate on-site construction.

Thus the challenge remains, to develop better models for the design and prediction of ponds' behaviour and effluent quality. This will continue to be the incentive for inquiring scientific minds to study, analyse and attempt to improve on previous studies.

Finally, if the application of the Fares' numerical model for wastewater treatment lagoons is a success story then it would, at last, provide a 'rational' design tool by which engineers may with confidence design systems for real retention times under local conditions almost anywhere in the world.

It is the combined use of the tracer study and the Fares' model that is important. The numerical model on its own would not be useful unless there was empirical hydraulic data to confirm that it is giving the right answer.
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APPENDICES
I TO XI
MATERIAL REDACTED AT REQUEST OF UNIVERSITY
APPENDIX 1 X

PREPARATION OF MEDIA FOR PHAGE EXPERIMENT

NUTRIENT AGAR

23 g Nutrient Agar
1 L distilled water

Method
1. Add agar to water and heat to just boiling with stirrer in order to dissolve.
2. Autoclave 15 mins @ 15lbs (121 °C).
3. Cool to barely touchable then pour plates.
4. Store in refrigerator @ 4-10°C.

BACTERIOPHAGE DILUENT

1 g Bacteriological Peptone
0.3 g Sodium chloride
1 ml Magnesium sulphate 0.5M
10 ml Tris hydrochloride 1M
Make up to 1 L with distilled water

Method
1. To make 0.5M magnesium sulphate solution: 3.1g MgSO₄ + 25 ml distilled water.
2. To make 1M tris hydrochloride: 15.959g + 100ml distilled water.
3. Add all components to 1 L volumetric flask and make up to 1 L with distilled water.
4. Adjust pH to 7.8 with 5N NaOH.
5. Dispense into test tubes with dispenser 9 ml.
6. Cap and autoclave @ 15lbs (121 °C) for 15 mins.

SOFT AGAR

9.1 g Bacteriological agar
11.2 g Nutrient broth
7.0 g Sodium chloride
1 L Distilled water

Method
1. Mix and heat with stirrer until dissolved.
2. Distribute into small bottles ~200ml.
3. Autoclave @ 15lbs (121 °C) for 15 mins.

NUTRIENT BROTH

8 g Nutrient broth
1 L Distilled water

Method
1. Mix and heat with stirrer until dissolved.
2. Distribute in 10 ml potions in test tubes.
3. Autoclave @ 15lbs (121 °C) for 15 mins.

PROCEDURE FOR GROWING PHAGE

A. Grow Host Organism

1. Subculture host onto nutrient agar plates.
2. Incubate plates @ 30-35°C overnight.
3. Check purity of host cultures.
4. Repeat purification process if necessary.
5. Incubate one colony into nutrient broth and incubate overnight.

B. Grow Phage Stock

6. Prepare a range of serial dilutions of the phage suspension supplied:

7. Melt some sterile soft agar, cool and hold @ 47°C in a water bath.
8. Label plates with appropriate dilution (as in diagram above) and volume sampled.
9. Add 0.5 ml phage from each test tube to plate with the same dilution.
10. Add 0.5 ml host bacteria broth to each plate and swirl to mix.
11. Pipette 3 ml of molten agar onto each plate and swirl to mix and distribute evenly across the plate.
12. Plates may be gently flamed to pop any bubbles produced.
13. Allow to set on a flat surface.
14. Invert plates and incubate overnight @ 30-35°C.
15. Inspect plates next day and select plates with almost confluent lysis.
16. Add 3 ml nutrient broth to each plate and allow to sit at room temperature for at least 20 minutes.
17. Collect broth into sterile glass bottles from plate surface using a sterile pipette.
18. Filter through 45μm membrane filter.
19. Then add 1 ml chloroform per 100ml phage stock to complete lysis of the bacterial cells. Centrifuge @ 3000rpm for 3 mins in order to separate chloroform. For harvested phage of smaller volumes use 0.5 ml chloroform.
20. This phage stock can be kept @ 4°C for many months without significant loss of titre.
21. Use 0.1 ml of this stock to ascertain titre.
## APPENDIX X

### pH, Temp and plaque forming units/ml (pfu/ml) PHAGE BENCH-SCALE EXPERIMENT - JUNE 93

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