RISK-BASED APPROACH TO ON-SITE WASTEWATER TREATMENT SYSTEM SITING DESIGN AND MANAGEMENT

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Abstract

The use of on-site wastewater treatment systems (OWTS) for the treatment and dispersal of domestic effluent is common in urban fringe areas which are not serviced by centralised wastewater collection systems. However, due to inappropriate siting and soil characteristics, the failure of these systems has become a common scenario. The current standards and guidelines adopted by many local authorities for assessing suitable site and soil conditions for OWTS are increasingly coming under scrutiny due to the public health and environmental impacts caused by poorly performing systems, in particular septic tank-soil adsorption systems. In order to achieve sustainable on-site wastewater treatment with minimal impacts on the environment and public health, more appropriate means of assessment are required.

The research described in the thesis details the processes adopted for the development and implementation of an integrated risk based approach to OWTS siting, design and management. This involved detailed investigations into resolution of some of the inherent deficiencies identified in the existing OWTS codes and guidelines, including more thorough site and soil assessment and data analysis, integration of the key risk facets of OWTS siting and design, environmental and public health, and the incorporation of scientific knowledge into the assessment processes. The research undertaken focused on four key research areas; (i) assessment of soil suitability for providing adequate treatment and dispersal of domestic wastewater; (ii) contamination of ground and surface waters as a result of OWTS failure and the major factors influencing contaminant fate and transport; (iii) assessment of environmental and public health risks due to poor OWTS performance; and (iv) the development of an integrated risk assessment framework for OWTS siting, design and management.

The research conducted was multidisciplinary in nature, with detailed investigations of the physical, chemical and biological processes involved in on-site wastewater treatment and dispersal. This involved extensive field investigations, sampling, laboratory testing and detailed data analysis across the fields of soil science, groundwater and surface water quality, chemical and microbiological contamination, and contaminant fate and transport processes. The interactions between these
different disciplines can be complex, resulting in large amounts of data being generated from the numerous field investigations and sampling processes undertaken. In order to understand the complex relationships that can occur, multivariate statistical techniques were utilised. The use of these techniques were extremely beneficial, as not only were the respective relationships between investigated parameters identified, but adequate decisions based on the respective correlations were formulated. This allowed a more appropriate assessment of the influential factors, and subsequently the inherent hazards related to OWTS, to be conducted.

The primary research objectives for this research were investigated through a series of scientific papers centred on these key research disciplines. The assessment of soil suitability was achieved through extensive soil sampling throughout the study area and detailed laboratory testing and data analysis. The studies undertaken are described in Chapters 3, 4 and 5. Paper 1 (Framework for soil suitability evaluation for sewage effluent renovation) outlines a framework for assessing the renovation ability of the major soil groups located throughout Southeast Queensland. This framework formed the basis for the assessment of OWTS siting and design risks employed in the developed risk framework. Paper 2 (Use of Chemometric Methods and Multicriteria Decision-Making for Site Selection for Sustainable On-site Sewage Effluent Disposal) details and justifies the multivariate data analysis techniques used in establishing the soil framework. Paper 3 (Assessment of soil suitability for effluent renovation using undisturbed soil columns) describes investigations of the use of undisturbed soil cores for the assessment of long term soil renovation ability. This study was undertaken to validate the research outcomes achieved through the established framework developed in Paper 1.

Papers 4, 5 and 6 (Chapters 6 - 8) focus on contamination issues of ground and surface waters resulting from poor OWTS treatment performance, and the different factors that influence pollutant fate and transport. The investigation of ground and surface water contamination, detailed in Paper 4 (Assessment of High Density of Onsite Wastewater Treatment Systems on a Shallow Groundwater Coastal Aquifer using PCA) and Paper 5 (Environmental and anthropogenic factors affecting fecal coliforms and E. coli in ground and surface waters in a coastal environment) was
achieved through extensive ground and surface water sampling and testing from several monitored study sites. Analysis of the resulting data indicated that several key factors, including rainfall, site and soil conditions and on-site system density can significantly influence the fate and transportation of pollutants emitted from OWTS. An additional issue found during this research in assessing faecal contamination of water resources was the necessity to ensure that the respective sources of contamination were actually OWTS. The inherent difficulty in identifying the actual source of contamination was resolved by employing a source tracking method, namely antibiotic resistance analysis of faecal coliforms (*Paper 6; Sourcing fecal pollution from onsite wastewater treatment systems in surface waters using antibiotic resistance analysis*). Finally, *Paper 7 (Integrated Risk Framework for On-site Wastewater Treatment Systems)* describes the development of the final generic integrated risk assessment framework and how the outcomes, as discussed through the previous 6 papers, were combined to assess the environmental and public health risks inherent in OWTS siting and design.

The outcomes of this research have significantly contributed to knowledge of best practice in OWTS siting, design and management. The developed soil suitability framework allows more appropriate assessment of soil characteristics for providing effluent renovation. This is generally not done in the current assessment techniques for OWTS. Additionally, the use of this framework incorporates scientific knowledge into the assessment of OWTS, allowing a more rigorous and scientifically robust assessment process. The processes and techniques used in the soil suitability framework, although based on the common soil types typical of South East Queensland, can be implemented in other regions, provided appropriate soil information is collected for the area of interest.

The integrated risk framework has also been developed on a generic level, allowing easy implementation into most assessment processes. This gives the framework the flexibility to be developed for other areas specifically targeting the most influential OWTS siting and design factors, and the potential environmental and public health hazards within those regions. The resulting research outcomes achieved through the studies undertaken were subsequently applied to a case study for the development of the integrated risk framework for the Gold Coast region. The developed framework,
based on scientific research, has allowed a more appropriate means of assessing site suitability for OWTS and appropriate management and mitigation of the environmental and public health risks inherent with poor OWTS performance.

**Keywords:** On-site wastewater treatment systems, risk assessment, risk management, multivariate analysis, groundwater, contamination
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The work contained in this thesis has not been previously submitted for a degree or diploma at any other higher education institutions to the best of my knowledge and belief. This thesis contains no material previously published or submitted for publication by another person except where due reference has been made.

Signed:                           Date:

(Steven Carroll)
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Dedication

I would like to dedicate this thesis to my father, Bernie, and to my mother, Gayle. Their patience and understanding throughout the completion of this doctoral research is gratefully appreciated.
CHAPTER 1.0

INTRODUCTION

1.1 Background to Research Project

On-site wastewater treatment system (OWTS) siting, design and management have traditionally been based on site specific conditions with little regard to the surrounding environment or the cumulative effect of other systems in the environment. Over the last few years, there has been increasing recognition that on-site wastewater systems are in fact treatment systems, providing a means of dispersing treated wastewater back to the environment or recycling it in a manner that protects both public health and the environment. This has led to the development of stringent treatment criteria and standards for on-site systems and the advancement of new technologies to meet environmental sensitivity (Siegrist et al 2000).

Consequently, this has lead to the recognition of two interrelated issues. Firstly, there is a need to ensure that appropriate technology is employed to adequately meet specific environmental safeguards. Secondly, the articulation of treatment standards and criteria which are flexible and robust enough to satisfy specific environmental requirements. This involves the integration of the concepts of hazard assessment and characterisation, risk assessment and risk management together with site specific characteristics in the siting, design and management of on-site sewage treatment systems (Jones et al 2000). The philosophical basis and language of risk are useful in that they provide a logical framework for considering methodology to meet criteria formulated to satisfy specific environmental standards. It enables a more rigorous, systematic decision making process than the performance based approaches currently being adopted (Hoover et al 1998, Eliasson et al 2001).

Unfortunately, these concepts are currently not being commonly applied to on-site wastewater treatment. Instead, stereotypical on-site system site evaluation and design criteria have been applied to sites located adjacent to environmentally sensitive areas
as have been applied to sites located in areas with less significant concerns. The general approach has been to apply the same framework of standards and regulations to all sites equally, regardless of the sensitivity, or lack thereof, to the receiving environment. Consequently, this has led to the continuing poor performance and failure of OWTS. Numerous incidence of poor treatment performance of on-site systems, in particular septic systems, is quite common (Harris 1995, Scandura and Sobsey 1997, Geary and Whitehead 2001, Lipp et al 2001, USEPA 2002). This situation is further compounded by the existence of large densities of such systems in many urban fringe areas, a common scenario which is particularly prevalent in the Southeast region of Queensland State due to rapid urbanisation.

Several studies have been conducted in Australia which all indicated that failure of OWTS is quite significant (Geary 1987, Jellife 1995, Goonetilleke et al 2000a,b, Goonetilleke et al 2002, Geary and Whitehead 2001). A recent study on the treatment performance of septic tanks within the Gold Coast City region indicated that approximately 90% of the investigated systems were not meeting the stipulated standards for effluent treatment (Goonetilleke et al 2002). Consequently, as a result of this study, the Gold Coast City Council considered it necessary to develop more suitable means of assessment and management of OWTS within their regulatory region. This in turn led to the current research project for developing a risk-based approach to on-site wastewater treatment system siting, design and management.

1.2 Research Aims

The main aim of this research was to develop an integrated risk-based approach for assessing and managing on-site wastewater treatment systems. This involved a multidisciplinary research approach to investigate the important aspects of the risk process, including assessing soil suitability for effluent renovation, fate and transport of pollutants from on-site system, microbiological assessment and soil and water chemical analysis. In summary, the primary aims of this research project were:

1. To develop a comprehensive understanding of the processes relating to on-site wastewater treatment systems, including site and soil characteristics and how these factors influence the effluent renovation process.
2. To develop a comprehensive understanding of the fate and transport of specific contaminants from on-site systems, including nutrients and pathogenic organisms.

3. To investigate the fate and transport of pathogenic organisms and appropriate means of sourcing their origin to improve the assessment of risk in relation to public health.

4. To develop a generically based integrated risk framework for assessing and managing on-site wastewater treatment systems.

5. To develop a GIS database to allow different risk characteristics to be analysed for their individual and cumulative impacts.

1.3 Scope and Research Objectives

On-site wastewater treatment systems are common throughout southeast Queensland. However, the concepts of risk assessment and risk management are currently not being applied for the assessment of risks involved in the poor performance of on-site systems. The main focus of this research was to incorporate strong scientific knowledge into an integrated risk assessment process to allow suitable management practices to be set in place to mitigate the inherent risks associated with on-site systems. To achieve this, research was undertaken focusing on three main aspects involved with the performance and management of OWTS.

Firstly, an investigation into the suitability of soil for providing appropriate effluent renovation was conducted. Although both subsurface and surface dispersal systems were investigated, the assessment of soil suitability was developed on the basis of effluent infiltration through the soil matrix. The research did not investigate the soil and site characteristics involved with surface or overland flow. The conducted research focused on the assessment of soil physico-chemical characteristics for attenuating and removing effluent contaminants as the effluent percolated through the soil matrix, as well as soil hydraulic characteristics necessary for effluent dispersal. This research was employed in the assessment of OWTS siting and design risks. Secondly, an assessment of the environmental and public health risks was performed specifically related the release of contaminants from OWTS. This involved detailed groundwater and surface water sampling and analysis to assess the
current and potential risks of contamination throughout the Gold Coast region. Finally, the outcomes of this research was utilised for the development of the integrated risk framework.

The conducted research was specifically formulated around the performance issues involved with the renovation of effluent after being discharged from on-site system treatment units. The treatment capabilities and performance of the actual treatment units and respective technologies was not investigated. Additionally, although specific management ideals were developed as part of the risk framework for mitigating the inherent risks investigated, it did not include general householder management practices, or development of public education programs.

Therefore, the primary objectives of the research project were:

1. To develop a methodology for hazard identification and characterisation, relating to on-site wastewater treatment in the context of the surrounding environment (groundwater, surface water and soil) and its sensitivity to the poor performance of on-site treatment systems;

2. To develop a cohesive integrated risk framework for the assessment of on-site system siting and design based on environmental and public health risk associated with on-site sewage treatment and to incorporate adequate scientific knowledge to reduce uncertainty in establishing identified risks;

3. To develop a critical point monitoring program to allow the continual monitoring and management of the environmental and public health risks associated with on-site systems and the continual refinement of the risk assessment process.

Although the main outcomes for the project were related to the Gold Coast area, the specific objectives, including the final integrated risk assessment framework, was aimed to be generic and applicable to other regions.
1.4 Justification of Research

OWTS are the most appropriate form of wastewater treatment in rapidly developing areas that do not have access to centralised treatment facilities. However, due to inadequacies in the current standards and codes utilised in assessing site suitability for the use of OWTS, poor performance is common. This ultimately leads to increased environmental and public health risks in areas that do not have suitable site and soil characteristics to prevent the release of poorly treated effluent from OWTS into the surrounding environment (Geary 1992, Siegrist et al 2000, Dawes and Goonetilleke 2004). In order to improve the general performance of OWTS, it is imperative that strong, scientifically based assessment and management guidelines are developed that are recognised across local regulatory boundaries. Unfortunately, this is currently lacking and existing codes and guidelines used for the assessment and management of OWTS can differ substantially from one jurisdiction to another.

The development of risk-based applications for OWTS was described as one of the single most important aspects lacking in the assessment and management of on-site systems in the United States at a recent National Research Needs Conference: Risk-based decision making for on-site wastewater treatment (Jones et al 2000, Nelson 2000). Although highlighting key research needs towards the assessment and management of on-site systems within the United States, similar issues are evident in other countries, including Australia, indicating that research conducted towards addressing these needs are of international significance. This research project was aimed at contributing to the knowledge base relating to the risk-based decision making associated with OWTS. In particular, the development of a risk assessment and management framework that is both universally acceptable and scientifically robust to reduce uncertainty in establishing the environmental and public health risks would be beneficial towards the assessment of OWTS and minimising the inherent risks associated with poor performance. This is particularly important for areas such as Gold Coast due to the current rate of development, high failure rates of OWTS, and the numerous environmentally sensitive areas for which the region is famous. The outcomes of this research, including the developed integrated risk assessment framework and subsequent risk maps will be a part of the development assessment process in the Gold Coast Region. This will allow adequate assessment of the
numerous OWTS within their regulatory region, and allow the management and mitigation of the identified risks.

1.5 Study Area

Research was confined to the Gold Coast region. Gold Coast region is situated in Southeast Queensland and covers approximately 1,500 km$^2$, bordered by Logan City and Brisbane City in the North, Beaudesert Shire to the west and the New South Wales (NSW) state boundary to the south, as depicted in Figure 1.1. Gold Coast region is a major tourist destination, with significant ecosystems such as, World Heritage sites, important water resources and Ramsar wetland sites located throughout the hinterland region. Additionally, the region is one of the most rapidly urbanising areas in Australia. Due to the escalating cost of infrastructure, on-site systems are the most economical and accepted means of wastewater treatment within the rapidly developing urban fringe areas. The region currently has over 15,000 on-site wastewater treatment systems with a majority of them being conventional septic tank-soil absorption systems. Large clusters of OWTS exist in various locations throughout the region as shown in Figure 1.1, and their cumulative effect has become a major concern for the Gold Coast City Council. Several areas have been identified as sensitive to environmental and public health impacts as a result of poor OWTS performance. These areas have high densities of OWTS (>500 systems/km$^2$) and commonly site and soil characteristics can be inadequate for appropriate effluent treatment and dispersal (Carroll and Goonetilleke 2004).

The Gold Coast region has a diverse range of soil types as shown in Figure 1.2. The most prominent soil is the Kurosol group, as classified under the Australian Soil Classification (Isbell 2002), which constitutes approximately 52% of the entire Gold Coast area. This is significant in relation to effluent treatment and dispersal, as Kurosol soils, formally categorised as podsolic soils and soloths, (including some red and yellow earths) under earlier soil classifications (Stace et al 1968) were considered inappropriate for effluent dispersal (Nobel 1996). The other problem soils with respect to OWTS are the Podosol and Tenosol soil groups, which constitute a majority of the soils along the coastal fringes. These are sandy soil, and although providing suitable dispersal properties, generally have minimal pollutant attenuation.
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and removal characteristics. The suitability of these sandy soils for effluent renovation is further reduced if they are influenced by seasonal or permanent shallow water tables. This particular soil group, classified as Hydrosols, are the least suitable for OWTS, providing minimal treatment, with dispersal directly into the groundwater itself. The remaining soil groups found in the Gold Coast hinterland region consist of a mixture of Ferrosols, Dermosols, and Kandosols soils with minor areas of Sodosols, Rudosols and Organosols. The Ferrosols, Dermosols and Kandosols are generally regarded as being suitable for effluent renovation.

**Figure 1.1:** Gold Coast City showing locations of OWTS
Figure 1.2: Soil association of Gold Coast City
1.6 Research Methodology

The implementation of a suitable methodology which adequately encompasses the specific research aims and objectives set out for this project was essential. The process of progressing from the initial problem formulation to the final integrated risk framework involved several iterations prior to achieving the specified objectives and development of the risk maps. Figure 1.3 illustrates the methodology adopted for this research project. Essentially, the methodology involved several stages to allow both the individual and final integrated risk frameworks to be developed via an iterative approach, which was refined with the collection of relevant data and progressive analysis undertaken. This process allowed the development of the risk frameworks to move from a qualitative approach based on empirical and qualitative relationships to a quantitative process incorporating appropriate scientific data and information.

The implementation of the developed research methodology was formulated around seven scientific research papers. Each of these studies focused on a specific stage of the research with the respective outcomes utilised in the development of the risk framework. The overall methodology and development of these scientific papers in the context of the research project is depicted in Figure 1.3. Detailed descriptions of the linkages between these scientific papers and their respective outcomes in relation to the development of the risk framework are provided in Section 1.8.

1.7 Multivariate Data Analytical Techniques

Soil and water analysis can be complex, resulting in large amounts of data being generated from the numerous field investigations and sampling processes undertaken through this research. This makes it difficult to manipulate or evaluate the resulting data on a univariate level due to the adverse relationships between analysed variables. To overcome this issue, the use of multivariate analytical techniques was adopted. Multivariate data analysis is beneficial in that large volumes of data can be processed for exploring and understanding relationships between different parameters. This is typically achieved through the procedures of pattern recognition, classification and prediction techniques.
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Figure 1.3: Schematic diagram showing detailed representation of research to be undertaken highlighting methodology and development of scientific papers.
Additionally, the use of multicriteria decision making aids, such as PROMETHEE and GAIA, allows the incorporation of selection, optimisation and decision making ordeals into multivariate processes. The use of these techniques were extremely beneficial, as not only were the respective relationships between investigated parameters identified, but adequate decisions based on these correlations were also formulated. The multivariate approaches utilised for assessing the data obtained through soil and water investigations consisted of three common methods.

**PCA**
PCA is a multivariate statistical data analysis technique which reduces a set of raw data into a number of principal components which retain the most variance within the original data to identify possible patterns or clusters between objects and variables. Detailed descriptions of PCA can be found elsewhere (Massart et al 1988, Adams 1995; Kokot et al 1998). After decomposition of the raw data matrix, principal components (PC’s) are chosen so that PC1 describes most of the data variance, followed by PC2 which retains the next largest amount of data variance but which is orthogonal to PC1. This means that PC2 is independent of PC1. The advantage of PCA is that most of the data variance is contained within the first few PC’s, reducing the dimensionality of the multivariate data matrix (Kokot et al 1998).

Objects (in this case soil and water samples) that retain similar variances in the analysed variables will have similar PCA scores which will cluster together when plotted. Likewise, relationships between variables can be easily identified by the respective coefficients. Strongly correlated variables will generally have the same magnitude and orientation when plotted, whereas uncorrelated variables are typically orthogonal (perpendicular) to each other. Clusters of object data and their respective relationships with the analysed variable can clearly be seen when respective scores and coefficients are plotted on a biplot. This allows respective relationships between analysed variables and respective objects to be identified.

**Discriminant Analysis**
To assess the ability of the different soil types for renovating effluent, discriminant analysis (DA) was employed to discriminate between major soil characteristics influencing the relevant processes. Discriminant analysis is a multivariate statistical analysis technique where a data set containing ‘X’ variable’s is separated into a
number of pre-defined groups using linear combinations of analysed variables. This allows analysis of their spatial relationships and identification of the respective discriminative variables for each group (Wilson 2002). Similar to PCA, objects that retain similar variances in the analysed variables will have similar discriminant scores and when plotted will cluster together. Similarly, strongly correlated variables will also have the same magnitude and orientation when plotted, whereas uncorrelated variables will be orthogonal (perpendicular). Visualising these biplots is undertaken in a similar manner as the well known PCA biplot.

There are two main functions for which DA is commonly employed. Firstly, it is used to analyse the differences between two or more groups of multivariate data using one or more discriminant functions in order to maximally separate the identified groups. Secondly, DA can be employed to obtain linear mathematical functions which can be used to classify the original data, or new, unclassified data, into the respective groups (Brereton 1990). Both techniques have been utilised through this research.

**PROMETHEE and GAIA**
PROMETHEE and GAIA are multicriteria decision making (MCDM) aids that rank actions according to specific criteria and thresholds. The details of PROMETHEE and GAIA are described elsewhere (Visual Decision Inc. 1999; Keller et al 1991), and therefore only a brief summary of the methods is provided here. The PROMETHEE method uses a pair-wise comparison system in which each action (soil sample) is compared to all other actions one-by-one defined by the preference functions, with thresholds and weights adopted by the decision-maker (Visual Decision Inc. 1999). PROMETHEE establishes preference flows ($\Phi$) for each action and ranks these based on the preference flows. Partial ranking (PROMETHEE I) utilises the $\Phi^+$ and $\Phi^-$ preference flows for ranking the actions. The positive flow, $\Phi^+$, determines the degree to which each soil sample is preferred over other samples, with higher positive values receiving a higher rank. The negative flow $\Phi^-$ determines the degree to which other soil samples are preferred over a particular sample. However, if samples have conflicting flows or preferences, they are considered incomparable in the PROMETHEE I ranking (Visual Decision Inc. 1999). The net
flow $\Phi$ ($\Phi = \Phi^+ - \Phi^-$), also called the $Pi$ score, represents the complete ranking ($PROMETHEE II$) of samples, with higher flow values ranked more highly. Both $PROMETHEE I$ and $II$ rankings were analysed to establish which soils were more suitable for effluent renovation.

GAIA provides a diagrammatic representation of the ranking methods of PROMETHEE, utilising a PCA technique. PCA is applied to the net preference flows ($\Phi$), and a biplot or GAIA plane, of the first two PCs is developed. Although no initial pre-treatment of data is needed to be undertaken, the preference functions established by PROMETHEE act to normalise the data, thereby providing some pre-treatment of the initial data. An additional feature of the GAIA plane is the incorporation of the $Pi$ decision axis. The orientation of the $Pi$ axis emphasises which criteria and actions are more dominant in the analysis (Visual Decision Inc. 1999).

1.8 Linkage of scientific papers

Satisfactory performance of on-site wastewater treatment systems, in particular septic systems, depends mainly on the ability of the underlying soil to renovate and transmit the discharged effluent. Consequently, one of the most important issues regarding the appropriate use of on-site wastewater treatment systems is the proper assessment of the site and soil characteristics which play a vital role in the treatment and dispersal of discharged effluent. However, past methods of assessing a soil’s ability to treat and disperse applied effluent have only relied on percolation tests and empirical relationships. Although these methods may indicate the soil’s ability to disperse effluent, they do not necessarily indicate the soil’s effluent treatment ability.

The concept of soil effluent renovation ability (ability of the soil to attenuate and remove pollutants and provide adequate dispersal) is described in detail through Papers 1, 2 and 3. Paper 1 ($Framework for soil suitability evaluation for sewage effluent renovation$) outlines a framework for assessing the renovation ability of the major soil groups located throughout Southeast Queensland, in particular the Gold Coast. Using multivariate statistical methods, the assessed soils were ranked in order of their ability to provide suitable effluent renovation based on their physical and chemical characteristics. The most significant outcome from this study was that
Kurosol soils, previously considered inappropriate for the renovation of effluent, did in fact provide sufficient renovation ability, and therefore were considered as suitable for on-site wastewater treatment. However, although this study provided the renovation suitability of major soil groups within Southeast Queensland, the suitability ratings were established based on scientific analysis of soils which had not been previously exposed to effluent. Consequently, further validation of these suitability rankings in relation to field conditions under long term effluent exposure was necessary.

*Paper 2 (Assessment of soil suitability for effluent renovation using undisturbed soil columns)* clarifies and confirms the outcomes of the soil renovation ability ranking described in *Paper 1*. *Paper 2* analyses the results of 8 months of laboratory column experiments using six undisturbed soil cores for providing adequate effluent renovation. The primary aim of this paper was to compare the original soil conditions, focusing on their physical and chemical characteristics, with that after 8 months of effluent application. Discriminant Analysis was undertaken to firstly discriminate between the different soil groups, and subsequently to assess the respective changes after effluent application. The outcomes of this investigation agreed with results of the developed framework, providing more scientific reliability as to which soils are more suitable for siting on-site systems and effluent treatment and dispersal.

Both *Papers 1 and 2* investigate the ability of common soil types found in Southeast Queensland for renovating effluent. However, although these papers use various multivariate statistical techniques to assess the soil physical and chemical data, a detailed description of the statistical methods adopted were not the main focus of these studies. Therefore, *Paper 3 (Use of chemometrics methods and multicriteria decision-making for site selection for sustainable onsite sewage effluent disposal)*, provides a more detailed discussion on the multivariate methods used for assessing site and soil suitability for effluent renovation, with a major focus on the multicriteria decision aids of PROMETHEE and GAIA. The combined outcomes from *Papers 1, 2 and 3* provided a more scientific basis for assessing the ability of soil to renovate effluent.
Although soil has a significant role in renovating effluent prior to discharging into water resources, inadequate assessment techniques have led to inappropriate soil types being approved for effluent treatment and dispersal. *Paper 4 (Assessment of high density of onsite wastewater treatment systems on a shallow groundwater aquifer using PCA)* highlights the need for better assessment and management of OWTS. Analysis of collected groundwater and surface water samples from 11 monitored locations showed that high densities of OWTS lead to significant nutrient and microbiological contamination of the shallow groundwater aquifer. However, it was also evident that some factors, such as soil and topography, climatic conditions and the type of on-site system used, play important roles which can either decrease or increase the extent of contamination.

The factors that influence the fate and transport processes of pollutants from OWTS are of significant importance, particularly in relation to the assessment of public health risks. The main focus of *Paper 5 (Environmental and anthropogenic factors affecting fecal coliforms and E. coli in ground and surface waters in a coastal environment)* was the assessment of various soil and site related factors that influence the fate and transport of microbiological contaminants into ground and surface waters near high densities of OWTS. The microbiological contamination of water resources is becoming important, particularly in light of the need to have adequate water supplies for rapidly developing areas. The research reports on the evaluation of extensive groundwater and surface water investigations undertaken and microbiological assessment through the use of indicator faecal coliforms and *E. coli*. Although several factors were observed to significantly impact on levels of faecal coliforms in the water, it was found that the combined impact of the investigated anthropogenic and environmental factors resulted in higher faecal pollution than any individual factor alone. However, the actual source of faecal pollution was not investigated in this paper.

Neglecting to identify the different sources of faecal bacteria has been a limitation of most studies assessing faecal contamination of water sources until recently. *Paper 6 (Sourcing fecal pollution from onsite wastewater treatment systems in surface waters using antibiotic resistance analysis)* seeks to address the issue of sourcing faecal indicator organisms. Whilst the presence of faecal bacteria in water resources does
indicate that faecal contamination has occurred, the faecal indicators may not be from one particular source, but rather from a variety of sources in the region, including on-site systems, domesticated and wild animals. Therefore, in order to reduce the uncertainty in assessing public health risks using faecal indicators in this research, Paper 6 describes the use of a Bacterial Source Tracking (BST) method, Antibiotic Resistance Analysis (ARA), to identify the respective source categories of the faecal coliforms obtained in collected water samples. Being able to accurately categorise the various sources of faecal contamination has a number of benefits. More suitable public health risk assessments can be achieved by removing a major uncertainty in using faecal coliforms for the assessment of public health. Additionally, identifying the major sources of faecal pollution allows more suitable management protocols to be implemented to control the respective sources of contamination. The use of ARA for this research project, as described in Paper 6, allowed the effect of faecal contamination from human sources, in particular OWTS, in several sensitive areas of the Gold Coast to be assessed. This information has since been used for assessing the respective risks to public health for the Gold Coast region and developing the risk assessment maps.

The development of the integrated risk assessment framework and how the outcomes, as discussed through the previous 6 papers, were combined to assess the respective risks and to identify the low and ‘at risk’ areas throughout Gold Coast region are discussed in Paper 7 (Integrated Risk Framework for Onsite Wastewater Treatment Systems). The developed risk-based approach shows that by developing the risk framework around the assessment process and relevant stakeholders, a more suitable framework can be developed with wide ownership among stakeholder groups, which can be successfully implemented into the current standards and guidelines.
CHAPTER 2.0
ON-SITE WASTEWATER TREATMENT

2.1 Introduction

On-site wastewater treatment systems (OWTS) are capable of providing adequate treatment and ultimate dispersal of domestic wastewater. However, the all too common ‘flush and forget’ attitude attributed by many householders using OWTS is a major problem. A current lack of knowledge and awareness regarding OWTS use, maintenance and management is all too common, particularly amongst householders. The type of system adopted is dependent on a number of siting, design, operational and management issues that need to be addressed in order to obtain adequate treatment performance. Although some regulatory control is evident, siting and design factors are still major causes of poor system performance. The operational and maintenance aspects are generally left to the householder. As such, regular operational and maintenance procedures are often neglected, leading to poor system performance and subsequent failure.

Numerous cases of system failure have been reported over the past years, increasing the concerns of regulatory authorities that on-site systems are not providing adequate treatment of domestic wastewater. The performance of on-site systems and the need to consider site and soil characteristics in their siting and design have been investigated by numerous researchers such as Brouwer and Bugeja (1983), Caldwell Connell Engineers (1986), Geary (1993), Geary et al (1999) Goonetilleke et al (2000). The outcomes of these studies confirm that many systems fail due to inadequate consideration of key site and soil characteristics, giving rise to potential environmental and public health risks. As such, it is recognised that a move away from the current performance based criteria used in the siting, design and management of on-site systems towards the evolving risk-based criteria is warranted. Subsequently, various risk-based approaches have recently been devised, ranging
from computer based models, to more refined management frameworks that incorporate various approaches aimed at reducing the inherent risks developed through the poor performance of OWTS.

This chapter firstly provides a review of the common forms of on-site wastewater treatment systems that are currently used for treatment of domestic wastewater in Australia in order to provide a comprehensive understanding of the processes involved in the wastewater treatment train. The issues and concerns dealing with system failure are reviewed, including a discussion on the transport and fate of the major contaminants of concern, including nitrogen, phosphorus and pathogenic organisms. The resultant hazards arising from the inadequate treatment of wastewater are also described. The concepts of risk, including risk assessment and management, and the various techniques commonly used to minimise the effects of risk exposure are reviewed. The applications of risk assessment and management to on-site wastewater treatment systems are described, and the current models, frameworks and methods used for assessing and managing OWTS are reviewed.

2.2 On-site Wastewater Treatment Systems

The need for on-site treatment systems is increasing at a significant rate due to new developments in urban and semi-urban areas which do not have access to centralised treatment plants and sewer systems. In the past, on-site systems were typically installed as temporary systems until centralised treatment systems could be implemented. However, this situation has not changed with on-site treatment systems becoming a permanent feature in most modern semi-urban developments. Approximately 25% of all housing units in the United States are connected to on-site treatment systems (Loomis et al 2001; Siegrist 2001), and this number is increasing every year. This trend is also evident in Australia where currently 17% of households employ on-site wastewater treatment systems (O'Keefe 2001).

In general terms, OWTS consist of three primary components; (1) the treatment unit; (2) the dispersal field; and (3) the soil (US EPA 2002). Typically, these systems can be broadly classified as either anaerobic systems or aerobic systems followed by a dispersal system, either subsurface or surface. The most common form of anaerobic
Chapter 2.0 On-site Wastewater Treatment

treatment system consists of a septic tank-soil absorption system. A typical setup for this type of system is shown in Figure 2.1. Various other effluent disposal options for septic tanks can also be utilised, such as mounds and evapotranspiration systems, where typical subsurface soil absorption systems are not suitable. However, Aerobic Wastewater Treatment Systems (AWTS) employing surface irrigation for final treatment and disposal of effluent are steadily becoming more popular, mainly due to their improved treatment performance. Sand filters have, of late, also had significant recognition in providing effluent polishing before ultimate disposal. However, sand filters are typically only employed where effluent quality is of significant importance in relation to the final disposal.

![Diagram of common on-site wastewater treatment systems used in Australia.](adapted from AS1547: 2000)

2.2.1 Septic Tanks

2.2.1.1 Overview

Septic tanks are the most common form of on-site wastewater treatment system used in Australia. Although often viewed as inferior to centralized wastewater treatment, septic systems are, and will continue to be the most prominent method of wastewater treatment in low density areas of development (Noss and Billa 1988). Septic tanks
allow the settlement of solids from the raw sewage, anaerobic digestion of organic matter, storage of sludge and scum, and discharge of clarified effluent for final treatment and dispersal (US EPA 1980, Goonetilleke et al 1999; USEPA 2002).

Septic tanks basically provide sufficient time for suspended solids and partially decomposed sludge to settle to the bottom of the tank and gradually accumulate. A layer of scum, composing of lightweight material, including fats and greases, rises to the surface of the clarified liquid (US EPA 1980). Regular removal of the accumulated septage approximately every three to five years is necessary to ensure that the system continues to function appropriately. Figure 2.2 shows a typical single chamber septic tank and the various zones developed within the treatment chamber.

![Figure 2.2: Typical single chamber septic tank, showing the developed scum, clear liquid and sludge layers.](image)

### 2.2.1.2 Operation and Maintenance

The proper operation and maintenance of septic tanks is crucial if they are to continually provide adequate sewage treatment. Hydraulic retention times are critical parameters that need to be considered in septic tank designs. The hydraulic retention time dictates the treatment process, defining how efficient the septic tank is at settling out suspended solids. The longer the retention time, the more time suspended solids have for settling, thus improving effluent quality. This provides a larger clear
water volume and accumulation of sludge, which in turn demands a large septic tank capacity (Goonetilleke et al 1999). Shorter retention times provide less time for suspended solids to settle, reducing the volume of clear water. This increases the level of sludge and scum accumulation in the tank, which in turn increases the level of solids in the effluent which is discharged to the soil absorption trench, thus accelerating clogging of the trench. There is general consensus that the most appropriate retention time suitable for adequate treatment performance is 24 hours (US EPA 1980, Bounds 1997, AS/NZS 1547: 2000).

Although septic tank systems are capable of providing suitable treatment of effluent, failure is a common. One of the most significant issues related to on-site systems and their poor performance record is that regular maintenance is not undertaken. Recent performance evaluations of on-site sewage treatment systems conducted in Queensland, Australia by Goonetilleke et al (2002), Goonetilleke et al (2000a,b), showed that the ‘failure’ rates of septic systems are directly related with poor maintenance regimes. Goonetilleke et al (2002) showed that 90% of septic systems investigated in the Gold Coast region exhibited poor treatment performance as sludge was not regularly removed. It is evident that householders have an inadequate awareness and lack of knowledge regarding appropriate maintenance and management of OWTS. In order to overcome the common “out of sight, out of mind” scenario, more effective education in relation to OWTS maintenance and management needs to be undertaken (Allee et al 2001).

### 2.2.2 Aerobic Wastewater Treatment Systems

#### 2.2.2.1 Overview

Aerobic Wastewater Treatment Systems (AWTS) can have significant benefits over the traditional septic tank-soil absorption systems, and are steadily gaining in popularity. Generally, AWTS are designed to replace septic tank-soil absorption systems where limitations relating to soil characteristics, topography and site conditions are not acceptable for the use of subsurface treatment and disposal systems. Aerobic (oxygen dependent) digestion of organic matter provides high quality effluents containing oxidised by products, carbon dioxide and metabolised
biomass (US EPA 1980, US EPA 2002). Oxygen is provided to the wastewater typically by mechanical means. The aerobic process differs from anaerobic processes where, through the presence of oxygen, organic material and ammonium-nitrogen are oxidised and digested by aerobic microorganisms. Suspended solids concentrations are also significantly reduced compared to anaerobic systems and, with the addition of disinfection devices, reduction in pathogenic organisms is also obtained (US EPA 1980).

Aerobic wastewater treatment systems generally consist of two main treatment processes. The first system type utilises fully aerobic processes, consisting of either suspended or fixed growth media for allowing aerobic bacteria to digest wastewater materials. Figures 2.3 and 2.4 provided typical configurations of fully aerobic systems. The second type of system is where an anaerobic chamber is employed as an initial pre-treatment process, as shown in Figure 2.5. The anaerobic treatment process always precedes the aerobic treatment. Each system has its own advantages and limitations, but in general, the same common features of oxygen transfer to the wastewater, contact between microorganisms and wastes, and solids separation and removal are utilised by each system (US EPA 2002).

Figure 2.3: Suspended growth aerobic treatment system. (adapted from Gustafson et al 2001)
From a wastewater treatment perspective, AWTS are beneficial for on-site treatment as they are capable of removing larger amounts of BOD and suspended solids than typical septic tanks. AWTS have also been shown to provide substantial nitrification.
of ammonia contained in the wastewater, and can also reduce pathogen numbers significantly (Brewer et al 1978; Hanna et al 1995).

Aerobic wastewater treatment systems generally employ fairly long hydraulic and solids retention times to ensure a high degree of treatment at minimal operational control (US EPA 1980). Although providing a higher level of BOD and suspended solids removal, high suspended solids washout can be a significant problem. As for septic tanks, some organic material will not be oxidized during the treatment process. Together with the long solid retention times adopted, these unoxidised materials will accumulate forming a sludge layer. This accumulation of sludge should be removed to prevent the washout of suspended solids. This is considered one of the main operational problems associated with AWTS.

Another significant disadvantage of AWTS is the nitrification of ammonia in the wastewater before ultimate disposal. This can be a major issue in regard to pollution of the surrounding environment and public health consequences (Converse and Tyler 1998). AWTS may contribute more nitrates to groundwater than a typical septic tank-soil absorption system due to the ammonia and organic nitrogen being converted into nitrate (NO₃⁻) within the treatment unit and not in the soil matrix. This can cause increased levels of nitrates in the soil, which can freely migrate to the groundwater. A recent study undertaken by Carroll and Goonetilleke (2004) showed that relatively higher levels of nitrate are present in groundwater in areas that had a high number of aerobic systems compared to areas that relied on septic systems. Other disadvantages associated with AWTS are:

- increased susceptibility to shock loadings, due to sudden high loading or intermittent loading;
- sludge bulking and periodic solids washout, which causes high variations in effluent quality; and
- larger volume of sludge produced when compared to anaerobic systems.

(Bailey and Wallman 1971; Otis et al 1974)
2.2.2.2 Operation and Maintenance

An important issue relating to the operation of aerobic wastewater treatment systems, as compared to septic tanks, is the mechanical devices employed to achieve aeration and recycling of biomass. These can have implications on the performance of AWTS, particularly through the malfunction of mechanical components or improper maintenance and design procedures (Hanna et al 1995, Khalife and Dharmappa 1996, Beavers et al 1999). Generally, AWTS manufacturers are required to undertake maintenance associated with mechanical malfunctions and equipment to ensure the system is operating correctly (in Australia this occurs approximately every three months). However, as with septic tanks, regular routine maintenance is left to the householder, who, in most cases, is not adequately trained to undertake the necessary maintenance required.

In general, AWTS are capable of providing high quality treatment of sewage, provided they are properly maintained. Khalife and Dharmappa (1996) have made the following recommendations to overcome the deficiencies relating to AWTS:

- The facility should be user friendly and simple to operate;
- Improved operation and maintenance is critical
- Homeowner education is necessary
- Training of regulatory staff in local authorities on technical aspects.

Inadequate frequency of the removal of suspended solids in the settling tank, which leads to solids carry over, is also a problem. Again, this is due to the lack of proper maintenance by the householder. Routine maintenance is essential for continual successful operation of AWTS. Unfortunately, regular maintenance is all too often neglected. Disinfection of effluent is typically employed for the effluent from AWTS, providing a much higher effluent quality, thus allowing reuse of the wastewater. Disinfection is commonly achieved by means of chlorine tablets placed in the system via a feeder, and dispensed into the effluent by dissolving in the wastewater. Although beneficial in disinfecting the effluent against pathogens, these devices have their limitations. These include:

- Masking of pathogenic organisms by suspended solids;
• Contact time with the tablet may not be enough, due to problems associated with the devices, such as feeder blockage; and
• Tablets missed either due to forgetfulness on the part of the householder or blocked feeders.

2.2.3 Other On-site System Alternatives

Although AWTS and septic tank systems are the most common system types utilised in Australia, other alternative systems types are available. These include additional treatment following conventional OWTS (nominally septic tank systems) such as sand filters, wetlands, media filters and a variety of alternative dispersal methods such as mounds and surface irrigation systems. On the other hand, alternative stand-alone treatment systems such as waterless toilets, using composting or incineration as disposal methods, and variations and adaptations of current system technology are increasing in popularity. These system types are generally utilised for various purposes, including improved effluent quality compared to current systems; reduced water usage, or to allow suitable recycling of treated wastewater.

2.2.4 Summary of Key Research Literature Findings

The review of current research literature has indicated that several alternative wastewater treatment systems are available for the treatment and dispersal of domestic wastewater. However, the most common system types currently utilised in Australia are the septic and aerobic wastewater treatment systems (AWTS). Although these systems are capable of providing suitable treatment of wastewater, a number of issues have been discussed throughout the reviewed literature that can highly influence the overall treatment performance. The two most common issues involve appropriate operation and maintenance of utilised systems and effluent quality. For any OWTS, undertaking regular maintenance is essential in order to ensure acceptable treatment of effluent is obtained prior to discharging to the dispersal field. Inappropriate maintenance results in poor effluent quality that can eventually lead to failure of the dispersal field, and subsequent environmental and public health risks.


2.3 Effluent Disposal

2.3.1 Subsurface disposal

Until recently, the final discharge of treated effluent from on-site wastewater treatment systems has generally been regarded as ultimate disposal. However, there is increasing awareness that the disposal of partially treated effluent to surface or subsurface environments is in fact part of the treatment process. Subsurface wastewater treatment and dispersal refers to the application of partially treated wastewater to a subsurface environment, with infiltration and percolation through the vadose zone (unsaturated zone), and finally into the saturated soil and underlying groundwater (Siegrist et al 2000). Figure 2.6 shows the major treatment components and effluent pathways associated with the subsurface dispersal process. The vadose zone is the final buffer between ground water and the contaminants contained in effluent applied to the soil. The depth of the soil vadose zone to ground water can affect hydraulic function, and in turn purification, by influencing the soil water content, aeration status, media surface area as well as hydraulic retention time (Van Cuyk et al 2001). Typically, a depth of at least 1m is regarded as appropriate for sufficient treatment of effluent (US EPA 1980, Siegrist et al 2000, US EPA 2002).

![Figure 2.6: Major components and pathways in subsurface disposal of effluent.](adapted from Bouma et al 1972)
It is only in recent times that factors relating to the soil and site conditions where treatment systems are installed have become important with regard to system siting and design. Previously, designs were reliant on simple percolation tests to evaluate site suitability. However, in recent years, researchers have shown that the percolation rate can be misleading and does not directly measure any soil characteristic that could be used in the design of the subsurface disposal system (Healy and Laak 1974, Gunn 1988). Consequently, site and soil related factors in the siting and design process gaining significant importance. The various soil and site factors that play an important role in selecting an appropriate system include:

- topographic considerations, such as site elevation and slope;
- subsurface considerations, including soil characteristics and profile, groundwater pathways, water table depth and variability and the depth to the limiting restrictive soil layer;
- area available for treatment and disposal;
- climatic conditions, such as rainfall and temperature;
- flooding frequency; and
- presence, location and distance to specific topographic features, such as waterways or wells.


In order for treatment and disposal systems to accommodate the long term acceptance of effluent, it is crucial that these factors are considered in the design and siting. In determining site suitability for OWTS, understanding the soil’s ability to accept, treat and disperse discharged effluent is crucial. Due to its heterogeneous nature, the assessment of a single soil parameter cannot provide a comprehensive overview of its suitability for a particular purpose (Diack and Stott 2001). As an example, the simple soil permeability test traditionally used as a means of assessment for effluent disposal, will indicate the soil’s ability to disperse effluent, but will not show if the effluent will undergo sufficient treatment prior to percolating into the groundwater. Therefore there is a crucial need for more scientifically rigorous procedure for assessing soil suitability for sewage effluent renovation and the removal of important pollutants.
2.3.1.1 Subsurface systems

The most common forms of subsurface treatment and dispersal systems in use in Australia are soil adsorption trench or bed systems. Trenches are constructed as shallow excavations with a single perforated pipe laid over gravel to evenly distribute the applied effluent (US EPA 1980). A typical trench system is shown in Figure 2.7. Soil absorption trenches are suitable where the soils are moderately permeable and remain unsaturated for a reasonable depth below the surface (Goonetilleke et al 1999).

![Figure 2.7: Typical trench system used for effluent treatment and ultimate disposal.](adapted from AS1547: 2000)

Bed systems differ from trenches in that the use of more than one effluent distribution pipe is provided over a wider area. A typical bed system is shown in Figure 2.8. This distribution setup results in a smaller effective area necessary for effluent distribution than that required for trench systems, making bed systems more
suitable for sites with restricted area. However, the same issue regarding treatment ability also applies to bed systems.

Several other subsurface system alternatives are available for use in areas where the typical trench and beds systems are considered inappropriate for providing adequate treatment and dispersal of discharged effluent due to restrictions in site and soil characteristics. These include:

- **Mound systems**
  Mound Systems are designed to overcome problems of treating and dispersing partially treated effluent in areas where the soil has a relatively low permeability or high ground water table, or where slowly permeable subsoils or subsoils overlying cracked bedrock exist (Bouma et al 1972; Magdoff et al 1974). The general mound system consists of an above grade soil adsorption system which relies on selected sand fill and top soil layers to purify the discharged effluent (Converse and Tyler 1984, 1987, 1998).

- **Evapotranspiration systems**
  Evapotranspiration systems utilise the natural climatic conditions to evaporate effluent from shallow trenches, combined with transpiration through the use of vegetation specifically planted to utilise the available water and nutrients. However, these systems are directly related to the climatic conditions. Evapotranspiration Systems are receiving increasing recognition in Australia, primarily due to its favourable climatic conditions which are necessary for the treatment and disposal of effluent.

### 2.3.2 Surface Disposal

Surface disposal, also termed land application, of pretreated effluent refers to the controlled application of effluent onto the land. This typically applies to effluent from aerobic wastewater treatment systems, which is applied to the land surface in order to achieve a further degree of purification and disposal. Surface disposal of effluent is achieved via two main methods, either by spray irrigation through the use
of a sprinkler, or via shallow drip irrigation of effluent just below the soil surface. The treatment involves similar purification mechanisms as for subsurface disposal. However, the excess nutrients in the effluent supplied to the land surface allow rapid vegetative growth, which aids in the treatment process. As the effluent passes through the soil-plant matrix, the effluent is treated in two basic ways. A portion of the effluent percolates through the soil, is purified and returns to the groundwater. The remaining effluent is either evaporated or is used by the vegetation, which transpire the wastewater to the atmosphere.

### 2.3.2.1 Implications

Traditionally, the design of effluent irrigation schemes has been based on the hydraulic criteria, disregarding the soil chemistry, leachate and groundwater effects (Balkau and Evans 1981). However, with the current increase in awareness relating to public health and the environment, the design and operation of surface application systems has come under scrutiny. In case of surface effluent disposal, irrigation of lawns and gardens can be considered as the most feasible (Goonetilleke et al 1999). However, various concerns relating to the use of surface application need to be considered. The main concerns involving the land application of effluent include:

- Land requirements
- Operation and maintenance; and
- Public Health and Environmental concerns


### Land Requirement

Surface application of wastewater requires relatively large areas of land in order to provide acceptable treatment quality and ultimate disposal. This is primarily related to the application rates for the effluent, which in turn can be dependent upon several other interrelated factors, including temperature, precipitation, evaporation, vegetation, subsurface characteristics, and slope (US EPA 1981). Neglect of these factors can result in hydraulic overloading and surface runoff could occur leading to environmental and public health concerns.
Operation and Maintenance

The operation and maintenance of surface application systems can play a vital role in the satisfactory treatment and disposal of effluent. Poor operation and maintenance practices can invariably increase the risks associated with public health and the environment. Proper disposal of effluent is dependent on two main factors. The quality of the effluent, and the method or operation of the system used to apply the effluent. Khalifée and Dharmappa (1996) in an evaluation of 27 surface disposal systems in Campbelltown City Council region in NSW found that only five systems, or 19%, functioned appropriately. Common problems noted by Khalifée and Dharmappa (1996) were:

- Inadequate land area and landscaping used for irrigation leading to hydraulic overloading of the application area;
- Application area being used for recreation and/or accessed by vehicles or livestock;
- Use of improper spray heads, causing erratic spray heights, increasing the likelihood of aerial drift; and
- Blocked spray heads by solids accumulation.

It is evident from the problems relating to the design, operation and maintenance of surface disposal systems that stringent regulatory procedures need to be implemented to safeguard against public health and environmental impacts.

Public Health and Environmental concerns

Public health is the most important consideration with regard to surface disposal of effluent. This is primarily related to pathogens and the transmission of disease through direct contact, inhalation of spray mist, and indirect spreading through food crops and potable water (Balkau and Evans 1981). Research into the fate of pathogenic organisms in relation to sewage disposal have shown that certain pathogens can survive for extended periods of time depending on the ambient conditions. Scherer (1982) has outlined the major environmental factors that affect survival of pathogen organisms. These are:
• Relative humidity - high humidity reduces droplet evaporation and organism die off, allowing organisms to survive for longer periods.
• Wind Speed - increases transport of pathogens through air
• Sunlight - Ultra violet radiation from the sun will kill off microorganism
• Temperature - increases in temperature can reduce the viability of organisms in the air.

The issues relating to public health can mostly be overcome if the effluent is satisfactorily disinfected (Goonetilleke et al 1999). Unfortunately, the incorporation of disinfection, typically through the chlorination of effluent, as part of the effluent treatment process is generally only utilised for AWTS. However, without adequate maintenance and management, the disinfection process may have limited success in achieving satisfactory treatment. This is related to the unreliability of the disinfection process used and the wastewater treatment quality and treatment processes available.

Land application of effluent can also have detrimental impacts on the environment. This is related primarily to the movement of nutrients (particularly nitrogen and phosphorus) either into groundwater or surface water. These consequences are influenced by factors such as insufficient land application area, high loading rates, slope and consequently runoff potential in the surface application area (Kleene et al 1993). However, with suitable management and design of land application areas, surface irrigation can be a suitable means of treatment and dispersal of effluent from on-site systems (Kleene et al 1993).

### 2.3.3 Summary of Key Research Literature Findings

The dispersal area is one of the most important components in the on-site wastewater treatment train. Both subsurface and surface dispersal areas provide the final treatment and dispersal of discharged effluent. Consequently, it is essential that the dispersal area is adequately assessed to ensure that proper treatment and dispersal is achieved without causing environmental and public health issues. However, although the dispersal area is generally an acceptable means of providing suitable treatment and dispersal of discharged effluent, the review of current literature indicated that
several performance issues can significantly influence overall performance. For subsurface application, the main issues involve effluent quality from the on-site system itself. Poor quality effluent can cause clogging of the soil pores, resulting in hydraulic failure of the dispersal field. For surface irrigation systems, appropriate assessment and management including effluent quality, applicable land area and protection against environmental and public health impacts is needed. Therefore, in order to ensure that proper treatment and dispersal of discharged effluent is achieved, it is essential that these issues are adequately addressed.

2.4 Performance of On-site Systems

2.4.1 Performance Issues

The performance of on-site wastewater treatment systems imposes several critical issues on the surrounding environment and the community. Improper on-site system siting, design and operation can result in biological and chemical contamination of water sources (Hagedorn et al 1981, Nichols et al 1997, Stevik et al 1999). In order to address and manage the critical issues arising from the inadequate treatment performance of on-site systems, due consideration of the various causes of failure is essential in establishing suitable design and performance criteria. The issues that are of importance with regard to the inadequate performance of on-site wastewater treatment systems include:

- Soil’s ability to renovate discharged effluent;
- Contamination of the surrounding environment, including ground and surface water;
- Public health concerns;
- Maintenance and management of on-site wastewater treatment systems.

The soil area used for effluent dispersal is of critical importance in the treatment of discharged effluent. The performance of an on-site system is not only related to the level of treatment produced by the treatment unit (septic tanks or aerobic treatment systems), but is also dependent on the capability of the soil to treat the discharged effluent percolating through the soil matrix. The soil’s renovation ability is a major
limiting factor in relation to OWTS. It must be noted that, although the soil is expected to play a crucial role in the renovation of wastewater, it may have a limited capability to achieve this. A very old system, for example, may function effectively for the hydraulic processes of the disposal system, yet may only accomplish limited purification. Therefore, its performance with respect to final treatment can be viewed as inadequate (Siegrist et al 2000). It is therefore obvious that the subsurface soil characteristics also need investigation before implying that a particular system is appropriate for the specific site conditions.

Under suitable conditions, soil can be an effective effluent treatment medium, relying on physical, chemical and biological processes to treat and dispose of the applied effluent. Due to the presence of finer pores which provide relatively more contact surface area for percolating effluent, clayey soils are generally better suited for effluent treatment and disposal. However, both the amount and type of clay present has a major influence on the ability of the soil to treat the effluent. Generally, the purification potential increases as the clay content increases, but the probability of clogging is also increased (Bouma 1974). The amount of clay is also significant as higher percentages can be impermeable, preventing vertical infiltration of effluent. As the effluent applied to the soil infiltrates through the soil medium, physico-chemical and biological processes of sorption, filtration and microbiological decomposition help in purifying the partially treated effluent, before it reaches groundwater (Miller and Wolf 1975; Van Cuyk et al 2001).

The process of sorption, or the binding of one substance to another, occurs through one of three mechanisms:

1. **Absorption**: a substance is totally taken in by another by molecular or chemical attraction.
2. **Adsorption**: a substance is bound to the surface of another.
3. **Persorption**: adsorption of substances in pores only slightly wider than the diameter of the adsorbed molecule.

(Miller and Wolf 1975; Ellis 1973)
The sorption processes in soil are beneficial for the renovation of effluent, provided the soil profile is suitable. The soil’s pore spaces possess a great ability for sorption of suspended solids and dissolved substances due to the electrostatic charges and chemically active surfaces available. Therefore, soil adsorption ability is reliant most significantly on the type of soil and its respective cation exchange capacity (CEC), which in turn is dependent on the amount and type of clay present in the soil and the amount of organic matter. CEC is basically a measure of the soil particles ability to exchange cations with freely mobile cations added to the soil matrix, in this particular case those associated with percolating effluent (Borden and Giese 2001, Manahan 2000). Therefore, if the soil has a low CEC, it will not have a strong ability to adsorb pollutants, and will rely wholly on filtration and microbiological decomposition processes. On the other hand, soils with a high CEC will have excellent adsorption ability, and provide suitable effluent renovation, providing the effluent can physically percolate through the soil. Clayey soils generally retain greater sorption ability due to the smaller size of the clay particles, and hence provide a greater, more active adsorption area. Clay soil particles which have a coating of iron, aluminium and hydrous oxides have exceptional sorption ability. The electrostatic properties of soils with high clay content, as well as organics, provide a good CEC which is capable of sorbing ionic and biological material, commonly contained within the percolating effluent (Miller and Wolf 1975).

Additionally, the generic properties of the soil matrix provide an effective medium for filtering out solid material from wastewater. This includes suspended solids and organic material. Soil contains numerous microorganisms which multiply rapidly with the extra nutrients provided through the effluent. The abundance of microorganisms in the soil aid in the filtering process by removing organics and nutrients. As such, the filtration capacity of the soil not only serves to remove suspended solids, but also to retain microorganisms to provide the biological treatment necessary for both dissolved and suspended organics (Miller and Wolf 1975).

One of the important issues relating to the filtration ability of the soil is the formation of a clogging layer or biomat. The biomat is composed of microorganisms living off the organic material percolating through the soil. The formation of the biomat occurs
when bacterial growth and their by-products, and accumulated solids reduce soil pore diameters resulting in the reduction of the soil infiltration capacity. Soil absorption systems often fail hydraulically due to clogging and subsequent reduction in infiltration through the soil at the soil-gravel interface (Bishop and Logsdon 1981, Kristiansen 1981).

However, even though the formation of the biomat is considered a major problem for the soil adsorption area, it can also be beneficial. The infiltrative capacity of the clogging mat has been shown to reach an equilibrium state after a certain period of time (Allison 1947, Jones and Taylor 1965, Otis 1984). As such, effluent will still be able to seep through the layer, but at a much lower rate. The development of this zone can enhance purification by increasing bio-geochemical reactions within the zone, as well as creating unsaturated conditions beneath it due to the reduced permeability rate below the biomat.

2.4.2 On-site System density

The density of on-site systems (or number of systems per unit area) and the issue of whether higher densities of OWTS cause adverse environmental and public health impacts remains an issue of debate. Several studies have investigated water quality near higher densities of OWTS, but to date none have provided significant proof towards how many systems in a particular area will cause detrimental impacts. Perkins (1984) and Yates (1985) both investigated the effect of high OWTS densities on groundwater, and found that increased densities do increase the potential for groundwater contamination, particularly in relation to nitrogen. However, the effect of increased densities on nitrate contamination of groundwater is not a linear process (Perkins 1984).

There are several factors that influence the potential for contamination due to high densities of systems, including the soil type, the type of system used, depth to the water table, as well as the general performance of the systems themselves. Typically, areas of major concern are developments on sandy soils with limited renovation ability, with a shallow or perched groundwater aquifer (Lawrence et al 2001). A
more recent study undertaken by Borchardt et al (2003) between septic system density and infectious diseases in children found that an increase in gastrointestinal disease outbreaks where higher in areas that had higher septic system densities than those with lower densities. However, the main reason for the increased outbreaks was caused by surface exposure to effluent (from poorly functioning or non-maintained systems) rather than through consumption of contaminated water supplies (Borchardt et al 2003).

2.4.3 Failure Consequences

2.4.3.1 Failure

The *failure* of on-site wastewater treatment systems is an issue that needs adequate recognition in order to prevent the possible ecological and public health hazards that may develop. Unfortunately, due to the lack of management and experience in the siting, design and operation of on-site systems, *failure* has become all too common (Goonetilleke and Dawes 2001). Table 2.3 highlights the more common failure scenarios related to OWTS. On-site wastewater treatment systems have shown they are capable of adequately treating and disposing of effluent over a 15 to 20 year life time if they are correctly designed, installed and maintained (Hodges 2001, Sowards and Fimmel 1983). Unfortunately, the widely accepted “*flush and forget*” attitude has proven to be a major problem. Generally, operation and management of on-site wastewater treatment and disposal systems has often resulted in poor performance and disposal of effluent (Charles et al 2001). This is due in part to the traditional management and inconsistent monitoring and maintenance processes whereby numerous assumptions have been applied to the siting, design and installation of OWTS with little consideration given to the long-term performance of the system (Hoover 1998).

Proper performance of on-site wastewater treatment systems depends on the ability of the soil to renovate and transmit wastewater. *Failure* occurs if either of these functions is not adequately achieved (Reneau Jr. et al 1989). According to Kenway and Irvine (2001), of the 284,000 on-site sewage facilities that currently exist in NSW, approximately 50-90% have failed or are performing inefficiently. Research
studies undertaken by (Goonetilleke et al 2002; Goonetilleke et al 2000a,b) produced similar results with failure rates of 70-90% for on-site wastewater treatment systems located in Brisbane, Logan and the Gold Coast regions. According to the US EPA (1996), failing septic systems is the second leading cause of surface and groundwater pollution in the United States, and is also the most frequently reported cause of groundwater contamination (Hoxley and Dudding 1994, Nicosia et al 2001, Perkins 1984; US EPA 1996, Yates 1985). However, the definition of what actually constitutes failure has proven to be contentious (Dix and May 1997).

Table 2.3: Failure Scenarios associated with OWTS

<table>
<thead>
<tr>
<th>Failure Scenario</th>
<th>Resulting Consequences</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hydraulic Failure of OWTS</td>
<td>Sewage ponding on ground surface near subsurface system or leakage on slopes; sewage pipe blockage and backup into pipes and fixtures; Elevated nitrate levels in drinking water sources; taste or odour problems in drinking water caused by untreated, poorly treated, or partially treated wastewater; presence of toxic substances (e.g. solvents, cleaners) in water source; Algal blooms, high aquatic plant productivity, low dissolved oxygen concentrations in nearby freshwater and marine water bodies</td>
</tr>
<tr>
<td>Groundwater and surface water contamination with chemical pollutants</td>
<td></td>
</tr>
<tr>
<td>Microbial contamination of ground and surface water</td>
<td>Shellfish bed bacterial contamination; recreational areas contaminated due to high bacterial levels; contamination of down-gradient drinking water wells with faecal bacteria or viruses</td>
</tr>
</tbody>
</table>

The term failure associated with on-site wastewater treatment systems has been loosely used throughout literature to describe major faults associated with on-site systems, including effluent surfacing, as depicted in Figure 2.11 and odour or mechanical malfunctions. However, with the current move towards the adoption of risk-based assessment of OWTS, the definition of failure needs to be defined in terms of the resultant hazards and exposure scenarios.

In relation to ecological hazards, failure refers to the inability of on-site treatment systems to provide adequate treatment and disposal of sewage, resulting in the increased release of contaminants, causing adverse conditions in the environment, such as eutrophication of waterways. In similar circumstances, hazards relating to
public health concerns associate failure of OWTS with the consequences of increased pathogen numbers in public areas resulting from the inability of OWTS to provide satisfactory removal of pathogenic organisms. In relation to ecological hazards, failure refers to the inability of on-site treatment systems to provide adequate treatment and disposal of sewage, resulting in the increased release of contaminants, causing adverse conditions in the environment, such as eutrophication of waterways. In similar circumstances, hazards relating to public health concerns associate failure of OWTS with the consequences of increased pathogen numbers in public areas resulting from the inability of on-site treatment systems to provide satisfactory removal of pathogenic organisms. Similarly, the hazards created from the inadequate siting, design, operation and management of OWTS, such as hydraulic overloading, effluent surfacing and infrequent pump outs also bring to light the various failures that need to be adequately addressed. Any treatment system can be deemed as failing if it allows harmful pollutants to accumulate to dangerous levels in the receiving environment (Otis et al 1974).

Failure of on-site treatment facilities, as defined by the NSW SepticSafe program (Brown and Root Services 2001), occurs when an unacceptable level of contaminants is released from the facility, including the land application area, to either groundwater or surface water pathways in the natural environment. As such, the movement of contaminants from on-site facilities will constitute a hazard to downstream receptors when they build up to high levels (Brown and Root Services 2001). These hazards generally relate to concerns involving ecological and public health issues.
The reason for these different views on failure is that on-site treatment systems have generally been installed and maintained according to the specified performance based criteria without taking into account all possible scenarios and outcomes related to inadequate on-site treatment and disposal of sewage. The failure of on-site wastewater treatment systems to treat and dispose of wastewater safely is not due to the inherent shortcomings of the systems themselves, as they can be very safe and effective when properly utilised. Failure results more from the misapplication and misuse of the system itself (Otis et al. 1974). This is primarily a result of the lack of direct control and routine by the householder and also the lack of proper regulations for controlling on-site wastewater treatment systems (Butler and Payne 1995).

The risks emerging from the inherent hazards imposed by the failure of on-site systems need to be adequately assessed to allow appropriate management techniques to be adopted in order to control and minimise their impacts. Unfortunately, this has not been sufficiently addressed for on-site wastewater treatment systems. Some risk assessments have been achieved through a qualitative approach in order to provide a toolbox for assessing the risks associated with on-site systems. However, these do not have the scientific depth needed to minimise the uncertainty associated with risk assessment. As such, it is necessary for a scientific based quantitative risk assessment be undertaken for on-site systems, not only to incorporate the scientific data to reduce the uncertainty, but also to provide adequate recognition of the major environmental and public health implications that urgently need addressing.

With the evolving risk-based criteria, it is essential that these different terms of failure are recognised, and integrated accordingly. There is a definite link between risk and failure specifically relating to the hazards imposed from the various failure mechanisms of on-site treatment systems. It is therefore appropriate to characterise failure as a mechanism for inducing hazards, and therefore define failure in terms of the hazards resulting from inadequate operation and management of on-site wastewater treatment systems.
2.4.3.2 Public Health

Public health is the most important concern related to wastewater treatment. Public health is generally concerned with pathogens and the possibility of disease resulting from contact with pathogenic organisms. It is significant to note that the majority of pathogens that affect human health originate from human wastes. Illness caused by pathogenic organisms can occur from direct or indirect contact, mainly via contaminated water sources. Various contact mechanisms include:

- maintenance of on-site systems;
- surfacing of effluent at individual sites;
- recreational activities, such as swimming or boating in polluted waters;
- drinking or using water from wells, bores or other surface water; and
- eating food crops that have been irrigated with treated effluent.

There is a wide variety of pathogens that can be found in water supplies, including bacteria, viruses, protozoa and helminths (Lawrence et al. 2001). Typical pathogens related to wastewater are listed in Table 2.4. These can have varying impacts on the human population, ranging from diseases such as Typhoid and Dysentery, to more severe viral outbreaks, such as Hepatitis A. It has been documented that pathogens can survive for extended periods in groundwater, and can travel outside the prescribed buffer zones of the treatment systems. The consequence of this occurring can be severe, and even fatal. Yates (1985) noted that approximately 50% of waterborne disease outbreaks in the USA were a result of consumption of contaminated groundwater, with septic tanks reported as the most frequent cause of contamination. The US EPA has estimated that approximately 200,000 cases of illness and 10,000 cases of water body impairment are reported each year due to contamination, with septic tanks considered as the second main cause of contamination (US EPA 1996).

One of the recent Australian cases relating to public health issues resulting from the failure of on-site systems was the Ryan versus Great Lakes Shire Council court case (Ryan 1999). The consumption of contaminated oysters from Wallis Lake in NSW led to an outbreak of viral Hepatitis A. The contamination of oysters with Hepatitis A was linked to failing on-site wastewater treatment systems in the Lake’s vicinity.
(Ryan 1999). This underlies the important need to have a means of assessing the siting, design and management of on-site wastewater treatment systems in general, before the inherent risks are at a level that may cause severe consequences. Unfortunately, the processes and mechanisms for studying contaminant fate and transport have not received adequate attention, with few research studies being conducted in this area (Lipp et al 2001, Siegrist and Van Cuyk 2001).

Table 2.4: List of pathogens commonly found in wastewater that can cause illness (adapted from Lawrence et al 2001)

<table>
<thead>
<tr>
<th>Pathogen</th>
<th>Source</th>
<th>Disease</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Viruses</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hepatitis A virus</td>
<td>Human faeces</td>
<td>Infectious hepatitis</td>
</tr>
<tr>
<td>Polioviruses</td>
<td>Human faeces</td>
<td>Poliomyelitis (best controlled through vaccination)</td>
</tr>
<tr>
<td>Astrovirus, Calcivirus,</td>
<td>Human faeces</td>
<td>Diarrhoeal diseases</td>
</tr>
<tr>
<td>Rotavirus, Norwalk-type</td>
<td></td>
<td></td>
</tr>
<tr>
<td>viruses</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Coxsackieviruses and</td>
<td>Human faeces</td>
<td>Diarrhoeal diseases</td>
</tr>
<tr>
<td>Echoviruses</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Bacteria</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Campylobacter jejuni</em></td>
<td>Human and animal faeces</td>
<td>Diarrhoeal diseases</td>
</tr>
<tr>
<td>Enterohaemorrhagic E. coli 0157</td>
<td>Human and animal faeces</td>
<td>Hemorrhagic colitis</td>
</tr>
<tr>
<td>Enteroinvasive E. coli</td>
<td>Human faeces</td>
<td>Diarrhoeal diseases</td>
</tr>
<tr>
<td>Enteropathogenic E. coli</td>
<td>Human faeces</td>
<td>Diarrhoeal diseases</td>
</tr>
<tr>
<td>Enterotoxigenic E. coli</td>
<td>Human faeces</td>
<td>Diarrhoeal diseases</td>
</tr>
<tr>
<td><em>Salmonella typhi</em></td>
<td>Human faeces and urine</td>
<td>Typhoid fever</td>
</tr>
<tr>
<td>Shigellae spp.</td>
<td>Human faeces</td>
<td>Dysentery</td>
</tr>
<tr>
<td><em>Vibrio cholerae</em> 01</td>
<td>Human faeces</td>
<td>Cholera</td>
</tr>
<tr>
<td><strong>Protozoan Parasites</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Cryptosporidium</em> spp.</td>
<td>Human and animal faeces</td>
<td>Diarrhoeal diseases</td>
</tr>
<tr>
<td><em>Giardia lamblia</em></td>
<td>Human and animal faeces</td>
<td>Diarrhoeal diseases</td>
</tr>
</tbody>
</table>

In addition to the pathogenic contaminants, chemical contaminants may also cause specific public health issues. Specific to on-site wastewater treatment, nitrate is the most conspicuous contaminant due to its solubility in water, and also its relatively free movement through the subsoil. Nitrate is the most stable form of nitrogen where there is an adequate supply of oxygen. As such, the most efficient means of reducing contamination is dilution in the groundwater. However, nitrate contamination
problems may not become obvious immediately, and where large nitrate concentrations are discharged to the groundwater, long term impacts are a significant problem. The most significant public health concerns related to nitrogen is the development of *methaemaglobinamenia* (blue baby syndrome) in young children, development of carcinogenic nitrosamines (cancerous cells) and also cyanosis, a fatal animal disease (Bouwer and Idelovitch 1987). However, debate over whether nitrate is a public health concern remains an issue. Nitrate is considered as being a critical contaminant in relation to both environmental and public health impacts, and water quality standards set restrictive values for nitrate. However, whether nitrate can be considered as a critical contamination in relation to public health is debateable (Patterson 2003). For instance, a 10 mg/L of NO$_3^-$-N threshold is set for drinking water. However, common editable foods, such as lettuce, beetroot and spinach can have nitrate values in excess of 1000mg/L (L’Hirondel and L’Hirondel 2002). The link between nitrate and *methaemaglobinamenia* was first noted by Comly (1945) who reported on two cases in which he stated that *methaemaglobinamenia* may be associated with high nitrate levels in drinking water and gastrointestinal disturbance. Subsequently, it has since been found that high nitrate levels in drinking water are typically also associated with microbiological contamination of the water (Fan et al 1987). More recent research has shown that nitrate by itself is not a public health concern, but the onset of various illnesses linked with nitrate, such as nitrosamines and in particular *methaemaglobinamenia*, is associated with a number of factors that occur simultaneously in conjunction with the consumption of drinking water with high nitrate levels (Avery 1999, Fewtrell 2004).

**2.4.3.3 Environmental Issues**

The main issue regarding environmental impacts is the potential for eutrophication of waterways. Eutrophication is the process in which oxygen concentrations in surface water is depleted through the decomposition of the excess nutrients by bacteria and algae (Rubin and Carlile 1991). This deprives fish and other organisms of oxygen, resulting in increased mortality of organisms and unhealthy waterways. The elevated nutrient levels can stimulate favourable conditions for algae and other water organisms to rapidly multiply, resulting in massive blooms (Harman et al 1996). This
can be a public health issue in some cases, particularly if the algae are toxic to humans, such as the common blue green algae, or the more serious Pfiesteria, which produces a toxin which can cause brain damage and organ failure if inhaled.

Unfortunately, there have been limited studies undertaken to investigate the fate and transport of discharged effluent in groundwater. Limited studies have also been directed towards investigating the effect of contaminant transport between groundwater and surface water, which is of particular concern to coastal communities and areas that are environmentally sensitive (Harris 1995, Paul 1997, Paul 2000). In areas with shallow freshwater systems, groundwater may easily be contaminated from on-site systems which are typically installed close to the water table. This situation is compounded during periods of high rainfall, which may cause flooding resulting in desorption of contaminants due to saturated conditions, which will then travel through the soil into the groundwater. The transport of pollutants from on-site systems can also be a significant source of contamination to the marine environment, especially in areas of restricted circulation, such as an estuary or small embayment (Corbett et al 2001).

2.4.4 Fate and Transport of Contaminants

Effluent discharged from on-site wastewater treatment systems contains specific physical, chemical and biological constituents which can contaminate the surrounding environment if not reduced to a safe level. The risks imposed by the release of these contaminants are significant. Typically, the transport of pollutants from septic systems occurs via groundwater flow, due to the use of soil absorption systems as the final treatment and ultimate disposal medium for discharged effluent. The processes and mechanisms that the soil uses to purify effluent can achieve a relatively high level of purification, but it will not ensure the total removal of these contaminants. Therefore, it is inevitable that a certain concentration of nutrients and pathogens will reach groundwater, travelling along available pathways to the receptors at risk. However, if the soil absorption field fails, surfacing of effluent may occur, with fate and transportation of nutrients and pathogens reliant on surface and climatic conditions. Surface runoff will then inevitably be a major transportation
process for contaminants. Surface runoff is also the major transport process of contaminants from surface disposal systems. Typical pathways for contaminant transport related to the hazards imposed on public health and the surrounding environment are shown in Figure 2.12. The consequences of the use of contaminated water can range from individual or site specific concerns, such as isolated outbreaks of disease, to contamination problems affecting a whole community, such as major disease outbreaks. Numerous reports of disease outbreaks have been reported throughout the literature, covering both isolated events and severe outbreaks. For example, 781 persons attending a Washington County fair became ill after consuming beverages made using contaminated groundwater from a nearby well. It was determined that the groundwater was being polluted from a nearby septic system (Cliver 2000). The outbreak of Hepatitis A at Great Lakes, New South Wales (Ryan 1999), as a result of poorly maintained septic systems is another example of public health risks associated with on-site wastewater treatment systems.

![Figure 2.12: Typical pathways of contaminant transport related to hazards imposed by on-site wastewater treatment systems](image)

In order to apply a safety net around on-site wastewater treatment systems, set back distances are typically specified, where it is implied that the inherent risks imposed by pathogenic organisms and chemical contaminants that do travel further than these adopted distances are significantly lower. Direct correlation between reduced contaminant concentrations and pathogenic numbers with distance provide evidence of this fact (Arnade 1999). It is conventional practice to overcome the potential risk of groundwater contamination leading to deterioration of downstream environments and public health concerns from septic tank effluent by specifying a minimum setback distance between the disposal area and potable water supply (Beavers and
Typical specified distances are in the range of 15 – 30m for wells and 100m from a perennial stream or waterway. However, this is dependent on the soil conditions at each site, and therefore these distances can vary (Viraraghavan 1978, Beavers and Gardner 1993). Corbett et al (2002) noted in their study on nutrient concentrations from three on-site treatment systems, that by increasing the currently adopted setback distance of 23m by an extra 50m, a 50% reduction of remaining contaminants, specifically nitrogen and phosphorus was possible. This reduction would also be dependent on the soil conditions present at the site. However, set back distances have also proved ineffective in numerous cases. A study undertaken by Robertson et al (1991) on contaminant plumes in groundwater from two single family homes on shallow unconfined sand aquifers, found that a plume of NO$_3^-$ and Na$^+$ had travelled 150m away from the effluent discharge point. At a sampling well located two kilometres away, concentrations of nitrate and sodium were also found, although concentrations were only one quarter of the allowable drinking water standard. This does, however, highlight the fact that contaminants are capable of travelling much further than typically expected. With the environmental concerns relating to the release of nitrogen and phosphorus into groundwater and ultimately surface water, and public health aspects related to the release of pathogenic organisms, more detailed discussions of the fate and transport of these pollutants are given below.

### 2.4.4.1 Nitrogen

Effluent discharged from septic tanks commonly consists of 80% ammonium-N (NH$_4^+$-N) and 20% organic N (Canter and Knox 1991, Gold and Sims 2000). On-site treatment systems can typically remove around 20% of the nitrogen contained in the effluent depending on the specific site related factors, such as soil, topography and climate (Siegrist and Jenssen 1989). Anaerobic conditions generally prevail in septic tanks, which provide excellent conditions for organic-N to be transformed into NH$_4^+$-N. Once in the soil, naturally occurring bacteria use the available organic-N and NH$_4^+$-N, oxidising it to nitrites and nitrates. Nitrification is a two step process which requires sufficient oxygen (aerobic environment) in order to occur. The first step of nitrification involves autotrophic bacteria (Nitrosomas or Mixotrophic) which
convert ammonium to nitrite in the following manner (Van Loosdrecht and Jetten 1998):

\[ \text{NH}_4^+ + 1.5\text{O}_2 \rightarrow \text{NO}_2^- + 2\text{H}^+ + \text{H}_2\text{O} \] (2.1)

Following this, heterotrophic bacteria (Nitrobacter) transform the nitrites to nitrates (Van Loosdrecht and Jetten 1998):

\[ \text{NO}_2^- + 0.5\text{O}_2 \rightarrow \text{NO}_3^- \] (2.2)

Nitrites are easily oxidised to nitrates, and therefore are not typically found in large concentrations, particularly in ground and surface waters. Nitrates, on the other hand, are the most mobile form of nitrogen, and tend to move easily through the soil and groundwater. Nitrates are therefore one of the major contaminants associated with discharged effluent from on-site treatment systems, and are considered a major contributor to eutrophication due to its high mobility. In order to remove the nitrates, denitrification must occur. Denitrification will follow nitrification if an anaerobic environment is present where \( \text{NO}_3^- \) will replace \( \text{O}_2 \) as an electron receptor for facultative aerobic or straight anaerobic dentrifiers (Pseudomonas or Achromobacter), which converts nitrates to nitrogen gas (Van Loosdrecht and Jetten 1998):

\[ \text{Not Strictly Anaerobic : } \text{HNO}_2 + \text{NH}_2\text{OH} \rightarrow \text{NH}_3 \]  
\[ \text{Strictly Anaerobic Bacteria : } \text{N}_2\text{O} \rightarrow \text{N}_2 \] (2.3)

For denitrification to take place, a sufficient amount of carbon must be present. The carbon provides the electron donor required for the transformation to nitrogen gas. These processes are significant in removing the nitrates from the effluent. Unfortunately, this rarely occurs on a significant scale as although the soil environment provides adequate aerobic conditions for nitrification, denitrification is
difficult. Unless the soil is saturated for a long period, anaerobic conditions will generally not exist, except directly below the soil absorption system in the first few centimetres of soil. As such, excess amounts of nitrates are able to freely move through the soil structure into groundwater if the nitrates are not nitrified within this thin layer of soil. This is a major environmental concern relating to the discharge of effluent. In the case of aerobic wastewater treatment systems, nitrification occurs inside the aerobic unit itself. Therefore, discharged effluent already contains a high level of nitrates, which is a cause for concern. However, as effluent is surface irrigated, most effluent is treated in the top soil layer where most nutrients are available for uptake by vegetation and microorganisms.

### 2.4.4.2 Phosphorus

Concentrations of phosphorus typically found in sewage effluent (~ 25 mg/L) are far in excess of what is found in the natural environment. This is mainly due to the levels of phosphorus in detergents and other household cleaning products commonly disposed in wastewater. The amount of phosphorus (< 0.02 mg/L) required to stimulate algae growth in aquatic environments is also far below typical concentrations found in effluent (Robertson et al 1998, Hesketh and Brookes 2000). There are presently no drinking water standards or regulatory upper limits established for phosphate concentrations, primarily because phosphorus is not a direct public health threat (Gold and Sims 2000). However, phosphorus is well known to have many undesirable impacts on aquatic ecosystems, as it, along with nitrogen, contributes to the eutrophication of waterways, which in turn can have a significant impact on human and animal health (Gold and Sims 2000).

In general, the digestion processes established in on-site treatment systems convert most of the phosphorus into soluble orthophosphates. This form of phosphorus is therefore able to move through aqueous regions, such as saturated soil, and ground and surface waters. However, due to its high exchangeability and reactivity with soil and chemical particles, phosphorus is generally retained in the soil. Once released into the soil, phosphorus undergoes various physical and chemical reactions. Firstly, adsorption to available exchange sites on soil particles is readily achieved. Typically,
phosphorus undergoes two sorption processes; fast sorption and slow ionic reaction. Fast sorption processes are generally confined to the sorption of phosphorus ions to the surfaces of soil physical and chemical elements. Generally, this occurs on the clay and organic matter particles. The slower, ionic reactions between phosphorus and chemical elements, such as aluminium, iron and calcium also occur, typically below the surface of the individual particles (McGechan and Lewis 2002). The pH of the soil can influence these chemical processes significantly. Soils that have a lower pH generally have higher levels of aluminium and iron available which enhances these slower ionic reactions. Similarly for soils with higher pH, more calcium is generally available in the soil for these reactions to occur. However, even though phosphorus is relatively less mobile than nitrogen and is considered as not being a major concern, phosphorus is still able to move into groundwater, generally over a much longer time period than nitrogen. As such, it is important to consider phosphorus as a possible hazard to water quality.

### 2.4.4.3 Pathogens

As public health aspects are concerned with the contamination of potable water by bacteriological, viral and protozoan parasite contaminants, attention is needed relating to their fate and transport through the soil media and into groundwater. Unfortunately, this has not been adequately addressed to date (Scandura and Sobsey 1997). Deborde et al (1998) highlighted several reasons for this, including:

- Human bacterial viral sources are present in faecal waste only when the source population is infected, requiring frequent sampling of sources over long periods of times, such as individual septic systems;
- There are many different types of pathogens that pose significant risk to public health;
- Assay techniques for human pathogenic viruses are complex, costly, and even non-existent for particular virus strains; and
- Injecting pathogens into an aquifer for research purposes, such as for researching fate and transport processes, is extremely difficult if not impossible.
The use of surface irrigation is also a significant concern, as the effluent is required to undergo disinfection prior to irrigation. This is generally achieved via chlorination of the effluent. As such, provided the effluent has been satisfactorily disinfected, the release of pathogens via surface irrigation will be minimal. However, if disinfection is not achieved, potential issues related to public health are raised. The risk of pathogens entering surface waters is a concern, mainly in relation to overland flow. In periods of dry weather, sunlight can effectively irradiate any pathogens escaping the disinfection process. However, periods of high rain pose the greatest concern.

Relating to subsurface soil treatment of effluent, pathogenic organisms are removed by one of four mechanisms; filtration, adsorption, microbial interaction or survival time (Laak 1986). These processes are influenced by the various factors including soil texture, composition and organic levels, pH, moisture content, temperature and competition from other organisms (Miller and Wolf 1975). As such, the mobility of pathogens in soil is greatly influenced by the size and type of the given pathogen.

The main concern relating to pathogenic fate and transportation is the moisture condition of the soil. Where an adequate depth of soil is available which is not influenced by shallow groundwater conditions, the soil will provide a suitable medium for effluent purification. However, the inability of soil to treat effluent as it percolates through the soil during the wet season, or due to high ground water fluctuations, results in groundwater contamination, thus posing a health risk to those who use groundwater as a potable water supply (Arnade 1999). On-site treatment systems located in coastal communities, or in areas with fluctuating water tables or shallow ground are of prime concern. Unfortunately, limited work has been undertaken on near shore surface water where high densities of on-site systems are present (Lipp et al 2001).

Saturated conditions may be beneficial for denitrification, but the mechanisms for removing pathogens; filtration and absorption, is substantially limited. This allows pathogens to freely move through the soil and into the groundwater, eventually returning to surface water. As such, it is of critical importance to assess the suitability of sites for potential groundwater issues. This can also be problematic near
coastal regions, or in wet/dry seasons due to the fluctuations occurring in watertable levels or groundwater.

The presence of cracks caused by insects and mammals living in the soil or by the shrink/swell properties of the clay under changing moisture conditions in the soil, can also affect the transportation of microorganisms through the soil matrix. This will short-circuit the movement of effluent through the soil, without providing any treatment whatsoever.

Another important aspect related to the fate and transport of pathogens is the selection of an adequate indicator of faecal contamination that can successfully represent the pathogens of concern, and allow satisfactory analysis of their transport and fate in the environment. Currently, total coliforms, faecal coliforms, *E.coli* and enterococci are the primary indicators used in water quality and risk assessments of faecal contamination (Meays 2004). Due to the large number of viruses and bacteria that are present both in the wastewater and naturally occurring in the soil and water sources, it is not possible to reliably predict the fate and transport of pathogens in the subsurface environment by utilising a common faecal indicator (Yates 1985). Additionally, these common indicators are present in the intestines and faecal matter of all warm-blooded mammals, including wildlife, livestock and humans (Hagedorn 1999, Whitlock et al 2002, Meays 2004). Consequently, the source of most faecal contamination can be a cumulative contribution from a variety of source hosts, making it difficult to accurately assess the potential public health impact resulting from faecal contamination. Additionally, faecal bacteria and coliforms are not reliable indicators of viruses due to physical differences between bacteria and viruses (Deborde et al 1998). However, as faecal indicator bacteria are much easier and less costly to detect and enumerate than the pathogens themselves, their use as a means of assessing the faecal contamination will remain common.

Recently, the use of Bacterial Source Tracing (BST) methods have shown promising results in sourcing faecal contamination. Essentially, the various sources of faecal contamination are identified by comparing unknown sources of bacteria or viruses to a library of known sources (Hagedorn et al 1999, Wiggins 1999, Meays et al 2004). These methods make it possible to separate certain bacteriological and viral
microorganisms into host specific sources (such as human, domestic and wild animal), allowing a more accurate indication of the source of the faecal contamination. This information is extremely beneficial as with the identification of specific sources of faecal contamination, better management of these sources can be achieved. Additionally, more accurate risk assessments for public health and environmental hazards can be undertaken once the source of contamination is known.

Several attempts at BST methods have been trialed in recent years with some success (Hagedorn et al 1999, Meays et al 2004) These included the ratio of faecal coliform to faecal streptococci (Sinton et al 1993); and proportions of thermotolerant coliforms to faecal sterols (coprostanol and 24-ethylcoprostanol) (Leeming et al 1998). Recent BST methods have shown to provide more accurate species differentiation. The use of molecular methods such as genetic makeup profiles of specific bacteria isolates, including ribotyping (Parveen et al 1999); random amplified polymorphic DNA (RAPD) (Parveen et al 1999, Tynkkynen et al 1999); rep-PCR DNA extraction methods (Dombeck et al 2000) and reverse transcriptase PCR (Hager 2001), have successfully been used to differentiate sources of both bacteria and viruses. Unfortunately, the main disadvantage of molecular BST methods is the high cost and longer time period required to obtain the necessary information needed for differentiation. Additionally, the use of physiological characteristics of bacteria have been used in biochemical BST techniques, such as Antibiotic Resistance Patterns (ARP) in order to differentiate sources of faecal bacteria which have been successful at determining sources of faecal indicators (Whitlock et al 2002).

Although BST methods enable the identification of the various sources of faecal contamination, the ability to accurately predict the fate and transportation of pathogens is still lacking. As such, indicators are the only means to date that have been successful in providing information on pathogen fate and transportation. This can also be misleading as the fate and transport of bacteria, viruses and protozoans can be significantly different.
A. Fate and Transport of Bacteria
Due to the large number of naturally occurring microorganisms in soil, it is difficult to select a specific species as a standard indicator (Canter and Knox 1991). This is a main reason for the limited research on the transport of bacteria through the subsurface environment. Selected bacterial indicators may not provide an adequate representation of the pathogen of concern. Many microorganisms are needed to renovate the chemical contaminants discharged to the soil. However, it is the disease causing organisms which are of concern.

In principle, the transport and fate of microorganisms is dependent on the properties and mechanisms associated with the soil matrix to which effluent is applied. The depth of soil between the bottom of the absorption bed, the groundwater, and the soil’s physical and chemical makeup are the most important aspects for bacteria removal. The primary mechanisms for removal of these organisms are the adsorption of bacteria to soil particles and chemical constituents, removal by filtration and die off. Research literature has noted that a distance of at least 1.0m is needed to provide sufficient unsaturated soil depth for removal of bacteria (US EPA 1980, Parker and Carbon 1981, Jones et al 2000). Die-off is dependent primarily on the type of bacteria, as rates vary for different bacteria (Wellings et al 1974). Saturated conditions can also increase the survival times of bacteria. Field experiments using strains of antibiotic resistant Escherichia coli were conducted by Rahe et al (1978) to evaluate the effects of saturation of a typical absorption field and the transport of E. coli once saturated. Movement of the E. coli at rates of 1.5m/day were reported, with survival rates of up to 96 hours being evident. The primary reason for the high movement and survival times were linked to the saturated conditions of the soil. Therefore, it is evident that soils with higher water tables, or which are continually saturated are at a greater risk of contamination than those with adequate unsaturated soils.

The ability of soil to adsorb bacteria is affected by various factors. Firstly, the Cation Exchange Capacity (CEC) of the individual soil particles is important as these are the primary adsorption sites. As such, with clay being the most reactive constituent, the type of clay and the available surface charges they possess are significantly important. Organic matter also contains high CEC and electrostatic charges which
makes organic matter viable adsorption sites. However, the organic matter is typically contained in the surface soil or ‘A horizon’ which may be of little significance in shallow soils or if the discharge point of the effluent is below this horizon.

**B. Fate and Transport of Viruses**

Although bacteria and viruses pose significant potential public health risks, viruses differ from bacteria in one important aspect. They are not cells but ‘obligate intracellular parasites’ that are incapable of replication outside a host organism (Beavers and Gardner 1993). Viruses are also significantly smaller than bacteria cells, making filtration an impractical mechanism for removal. As such, the main mechanisms for virus removal are adsorption to the soil particles, and die-off of the viral cells. The mechanism of adsorption of virus cells is generally the same as for bacteria, primarily relating to the soils structure and constituents, with primary emphasis on the electrostatic charges of the individual soil particles. Virus transport in unsaturated porous media is distinguished from transport in saturated media, because sorption and inactivation are considerably influenced by soil moisture content and subsurface temperature fluctuations (Sim and Chrysikopoulos 2000). Consequently, the only mechanism left to prevent virus survival is die-off if conditions reduce the effectiveness of the mechanism of adsorption taking place, such as saturation, shallow ground water or just poor soil conditions. A study undertaken by Nicosia et al (2001) on two field sites for assessing the removal of bacteriophage PRD1 from septic tank effluent found specific fate and transport patterns in drain fields were significantly influenced by rainfall effects, which aided in desorption of viruses from the soil. They also noted that the main mechanisms contributing to virus attenuation included inactivation and dilution or dispersion.

The major variable in determining virus inactivation is time. Clearly, the longer a virus persists in the environment, the greater the risk of infection when it reaches groundwater (Beavers and Gardner 1993). It must be noted however, that the survival of virus in soil is directly related to the type of virus (Wellings et al 1974). Viruses have been found to persist in saturated conditions for up to 120 days, but typically virus survival times is generally in the range of 20-40 days (US EPA 1980).
Sandy soils adsorb viruses poorly, thus allowing extensive migration into shallow ground water (Scandura and Sobsey 1997). This also relates to how far a virus plume can be transported by groundwater flow, as the longer the survival time, the further the plume can travel. As viruses cannot reproduce outside a host, they are virtually not an issue until they infect a host. Therefore, unless contact with infected groundwater occurs, outbreaks will not take place.

C. Fate and Transport of Protozoan Parasites
Protozoan parasites pose a lower risk of polluting groundwater from on-site wastewater treatment systems, than bacteria or viruses. This is due to their relatively larger size compared to other pathogens (Lawrence et al 2001, Thurston et al 2001). The two most common protozoans that typically pose a health risk are Cryptosporidium parvum and Giardia Lamblia. However, most cases of infection are self limiting in healthy adults, which reduces the risk of major outbreaks (Lawrence et al 2001). Protozoa reproduce by releasing eggs, commonly referred to as oocysts (Cryptosporidium parvum) or cysts (Giardia Lamblia), which will survive for a significant time, and only hatch when in a host body. These oocysts and cysts are particularly robust, making disinfection of effluent ineffective for their removal (Gibson et al 1998).

The main mechanisms for removing protozoa are the same as for bacteria and viruses; filtration, adsorption and die-off. Die-off is of major concern due to the resilience of the oocysts and cysts. These can survive for extremely long periods of time, making die-off the least effective means of removal. Therefore filtration and adsorption are the effective processes for removal. Due to their relatively large size, filtration is very effective in removing both the adult parasite and the oocysts and cysts from wastewater. However, this is directly related to the grain size of the filtering media and the hydraulic loading rate (Harvey et al 1995, Logan et al 2001). Those that escape through the filtering media will generally be adsorbed by soil particles, making contamination of groundwater less significant. As such, it is the direct release of protozoans into surface waterways that poses the greatest risk of infection. This is important in surface irrigated effluent, or surfacing of effluent through failed systems, with surface runoff the main pathway of contamination.
2.5.5 Summary of Research Literature

The reviewed research literature has identified important issues that can significantly affect the overall performance of OWTS. These mostly involve appropriate assessment of the respective site and soil characteristics which determine the overall ability of the OWTS for providing appropriate treatment. Inadequate assessment can lead to poor performance and eventual failure of OWTS resulting in severe environmental and public health issues.

The review of research literature enabled understanding of contaminant fate and transport processes for the major contaminants associated with OWTS, including nutrients and micro-organisms. The mechanisms that influence the attenuation and removal of these contaminants are generally associated with the physical and chemical characteristics of the soil medium, including filtration, adsorption and die off (for micro-organisms). As such, it was clearly highlighted throughout the reviewed literature, the importance of ensuring adequate site and soil assessment in order to prevent poor system performance and the inherent environmental and public health impacts.

2.6 Risk and On-Site Wastewater Treatment

As outlined by the numerous issues affecting the siting, design and management of on-site wastewater treatment systems discussed in the previous sections, adequate recognition of the inherent risks imposed is needed. As such, the need to assess and manage the inherent risks associated with on-site systems is necessary to adequately safeguard environmental and public health values. However, in order to achieve this, a generic understanding of what represents both hazards and risk, and means of managing them is needed.
2.6.1 Generic Concepts of Hazard and Risk

2.6.1.1 Hazards

A hazard exists when there is potential for harm towards either human or environmental receptors. Specific to on-site wastewater treatment systems, hazards are primarily related to the release of pollutants into the receiving environment, as shown in Table 2.5. Therefore, the most important step in the overall risk assessment process is to identify all the potential hazards resulting from the occurrence from a particular event, such as the inadequate performance of an on-site wastewater treatment system. There may be varying degrees of hazards associated with different environmental and public health situations (Asante-Duah 1998). As such, all relative hazards need to be identified, no matter how big or small they may be in comparison. This results from the possibility of a cumulative effect which can pose quite significant and severe consequences from the diverse range of potential hazards.

Table 2.5: Hazards and contributing factors related to OWTS

<table>
<thead>
<tr>
<th>Item</th>
<th>Key Hazard</th>
<th>Contributing Factors</th>
</tr>
</thead>
</table>
| OWTS (Treatment system and disposal area) | Release of contaminants due to 'failure' of On-site wastewater treatment system | 1. Soil  
2. Planning (Lot size)  
3. Environmental Sensitivity  
4. Flooding  
5. Topography  
6. Loading rates  
7. Operation and maintenance practices |
| Surrounding Soil       | Inability to renovate effluent and prevent contaminants from reaching groundwater and/or surface water | 1. Soil Type  
2. Depth of soil horizons  
3. Physical characteristics  
4. Chemical characteristics  
5. Water table depth |
| Public Health          | Contamination of water/surrounding environment such that a considerable health risk is evident due to the release of contaminant (namely pathogens) which have an impact on human health | 1. Surface exposure  
2. Water supply (ground/surface)  
3. Aerosols  
4. Pests (mosquitoes etc) |
| Environmental          | Release of contaminants into the receiving environment (ground/surface waters) causing environmental degradation (such as eutrophication) causing the environment to be unsuitable. | 1. Surface runoff  
2. Groundwater discharge  
3. Flooding  
4. Water table |
One of the most important questions to ask when identifying hazards is “when do hazards actually represent a risk?”. In order answer this question, one must first understand the process that leads towards a hazard, and consequently, risk. Figure 2.13 represents the potential risk associated with hazards, exposure pathways and those at risk. Essentially, a hazard becomes a potential risk when the respective exposure points related to the specific hazard instigate potential consequences to a specific population either directly or through the related exposure pathways (Jones et al 2000). In relation to identifying a hazard, the exposure pathways also need to be identified. For a typical on-site septic tank-soil adsorption system, the exposure pathways for contamination representing specific hazard are shown in Figure 2.12. Once identified, the potential hazards relating to the exposure pathways can be characterised, and the populations at risk can be identified. A thorough risk assessment can then be carried out.

![Diagram](image)

**Figure 2.13:** When do hazards actually represent risks?  
(adapted from Asante-Duah 1998)

### 2.6.1.2 Risk

There is no universally accepted single definition of risk given in the literature that can be applied in specific terms relating to specific areas (Asante-Duah 1998). For instance, the definition of environmental risk will differ from that for public health risk, which will also be quite different to that for financial risk. However, in simple generic terms risk can be defined as:
“The ‘likelihood’ of a course of action (or lack of a course of action, as the case may be) will result in an event leading towards a potential for harm (hazard). Risk is measured in terms of the consequences arising from the event and its likelihood” (AS/NZS 4360: 1999; Jones et al 2000)

In other words, risk is a two dimensional entity involving the possibility of adverse consequences resulting from a hazardous event, and uncertainty (Covello and Merkhofer 1994). It must be noted that the concept of risk involves two specific elements:

- The frequency or probability (aka ‘likelihood’) that a hazardous event will take place; and
- The consequences of that event taking place.

As such, risk can be summarised as the product of these two components, or:

\[ \text{Risk} = p \times S \]

where \( p \) is the probability of an event occurring, and \( S \) represents the consequences of that event occurring. (Asante-Duah 1998).

One of the most fundamental concepts related to risk is that it can never be eliminated (Jones et al 2000). This is one of the most misunderstood factors related to risk. It is possible to reduce or minimise the risk through good management practice, but risk will always be present as long as potential hazards exist. Additionally, judgements about the likelihoods and consequences of such events depend on a broad range of uncertainties, most of which are unacknowledged in current approaches to risk assessment (Burgman 2001). As such, uncertainty is an entity that cannot be forgotten in evaluating risk, but adequate means of reducing the amount of uncertainty is essential. In evaluating uncertainty in a site-specific risk assessment, Labienice et al (1997) concluded that the uncertainty in site properties can substantially impact on the fate and transport of contaminants, and in turn can significantly alter the overall risk assessment process. This in turn will affect risk management procedures. As such, it is essential that the uncertainties involved in risk
be evaluated to understand how they will affect the likelihood and consequences of related hazards.

The process of assessing and managing risk consists of a series of major processes, as depicted in Figure 2.14, including (1) Problem formulation; (2) Hazard Identification; (3) Risk Assessment and (4) Risk Management (AS/NZS 4360: 1999; Asante-Duah 1998; Burgman 2001; Chin and Chittaluru 1994; Covello and Merkhofer 1994; Gough 2001)

![Figure 2.14: Major process involved with assessing and managing risk (adapted from AS4360:1999)](image)

These individual processes have various procedures that define a framework for implementing each step. These are discussed in the following sections. It must be noted however, that the concept of risk assessment and management is an iterative process. Achieving one item, such as successfully reducing a risk, may significantly impact in another area, and as such, continual monitoring and review of the risk assessment and management process needs to be undertaken.
Monitoring and review is of significant importance when assessing and managing cumulative risks. Cumulative risks are those which consist of a number of hazards, all combining to produce an adverse consequence at a particular location or point in time. It is important to accommodate cumulative effects, as Solomon and Sibley (2001) have noted. Contaminants seldom occur on an individual basis, with most hazardous events incorporating a wide variety of contaminants. The complexity and variety of contaminants usually evident in hazardous events can make assessment on an individual basis difficult (Solomon and Sibley 2001). As such, the ability to assess these issues on a cumulative level provides a convenient method for dealing with the associated risks. In the case of on-site wastewater treatment systems, the release of nutrients or pathogens into groundwater from individual systems produce risk related to individual systems. However, the combined affect of a cluster of systems give rise to cumulative impacts on the receiving environment. This cumulative effect will not however be evident until the consequences of the combined effect have become noticeable.

Additionally, the inclusion of appropriate stakeholders who have an essential role in the development of the risk assessment and management process is one of the essential elements that should form part of any risk management process (AS/NZS 4630. 1999). The inclusion of stakeholders in the overall risk management process is essential, as more often than not, the key stakeholders will be those who are most at risk. As such, the key stakeholders will need the opportunity to participate in the risk management process. However, it will not always be easy or possible to include all relevant stakeholders due to various issues that can impede the management process. This includes the fact that not all organisations will want to communicate with the public due to confidentiality issues, legal implications, and due to exposing the identification of interested parties and stakeholders. Also communication and consultation can be expensive and may not provide the details necessary to those involved (Gough 2001). These issues themselves bring forth other political, legal and financial risks that may not have even been considered in the initial risk management process.
2.6.1.3 Risk Assessment

Various definitions of risk assessment can be found throughout the literature (AS/NZS 4360: 1999; Asante-Duah 1998; Chin and Chittaluru 1994; Covello and Merkhofer 1994; Gupta et al 2002), but these have generally been defined around the risk assessment methodologies which various assessors have devised. In a generic sense, risk assessment can be defined as the scientific process of evaluating the possible likelihood and consequences of the specified risk (AS/NZS 4360: 1999; Chin and Chittaluru 1994). This entails both risk analysis (the use of collected information to identify possible hazards and estimate the risk imposed on the public or the environment) and risk evaluation (the process in which judgments are made on the level of risk, based on risk analysis of environmental, social and monetary factors).

In general, three distinct types of assessment methodologies are used; qualitative, quantitative or semi-quantitative risk assessments (AS/NZS 4360: 1999). Qualitative assessments involve ranking the identified hazards based on descriptive scales of risk, which can be adjusted or adapted to suit the circumstances (AS/NZS 4360: 1999). As such, there is no indication of the uncertainty involved in these rankings. Generally, the descriptive scale employed is based specifically on the assessors’ and stakeholders’ perception of the hazard. Contrary to this, quantitative risk assessment uses statistical analysis of collected scientific data to provide the likelihood (or probability) of the consequence of the hazard occurring. Outcomes of the analysis provide a means of ranking the levels of risk, with a reduced level of uncertainty involved due to the use of scientific data. However, this is strictly dependent on the data quality.

Semi-quantitative assessments use the same principles of quantitative risk assessments, although no explicit statistical analysis is used to provide the likelihood of occurrence. Scientific data may be used to rank the hazards on a numerical scale to provide a likelihood of the consequences, although the ranking is subjective to the assessor’s perception. This provides an appropriate rank which is used for assessment purposes. Unfortunately, the methodologies used for qualitative assessments do not adequately address the uncertainties surrounding many
assessments, primarily because they are unable to do this (Burgman 2001). This is specifically due to qualitative measures adopted for assessing the risks, rather than using scientific data to assess the probability or likelihood of the consequences.

There are several ways in which the principles of risk assessment can be incorporated into the field of on-site wastewater treatment. These include comparative, discipline-specific and integrated type risk assessments (Jones et al 2000). Comparative risk assessments are typically used to assess and decide among alternative courses of action, such as assessing the estimated risks from nutrient loadings from either centralized or decentralized treatment (Jones et al 2000). On the other hand, discipline-specific risk assessment processes are related to specific technical disciplines, addressing issues based only in specific fields of expertise. These would typically be used in assessing risks on a scientific basis, but not being concerned about ecological or public health disciplines. Finally, integrated risk assessments are more broadly targeted at gathering information from various sources, assessing the involved risks and combining them into a single cohesive framework. This framework is specifically intended to support integrated risk assessments such as assessments used for on-site wastewater treatment systems (Jones et al 2000).

**Risk Assessment Techniques**

Despite the common use of the term ‘Risk assessment’, considerable differences in the processes and relevant methodologies and frameworks used in assessing risk are noticeable (Bridges 2003). Consequently, the type of risk assessment technique to be employed during the risk analysis and evaluation phase is dependent on the type of assessment being conducted, the respective stakeholders, identified receptors and endpoints, and the amount and type of information available. For singular assessments such as public health or environmental risk assessments, more detailed analysis can be conducted, as more specific data relating the inherent hazards will be targeted. However, for integrated risk assessments, the combination of various data and influencing hazards may pose a significant challenge to undertake an extremely detailed risk assessment.

In assessing risk of public health hazards, it is common to either apply a probability distribution technique based on applied data, or to conduct a dose-response
relationship to determine at what dose of a particular substance or organism (e.g., pathogen) a specific response will be developed in identified receptors (e.g., humans). The dose-response relationship is currently reliant on a group of sigmoidal mathematical equations which are used to empirically describe the required dose-response relations (Buchanan et al. 2000). The level of risk is then assessed on this dose-response relationship. However, the use of the required dose-response model is dependent on the availability of information on the population’s exposure to the biological agents as well as an understanding of the mechanisms of pathogenicity (host, food, and type of pathogen) associated with individual pathogens (Buchanan et al. 2000, Ashbolt 2004).

Dose-response relationships have been developed for a majority of chemicals and microorganisms, and the risk assessment phase can be assessed based on these developed models. However, until recently, most utilised dose-response models have used data based on relationships developed through animal trials, which can instigate some uncertainty in their predictive capability (Ashbolt 2004). Similar techniques are utilised for ecological and environmental risk assessment, with the exception that the relevant receptors are typically non-human (risk assessment focused on consequences related to animals and environmental sensitive areas). However, although risk assessments this detailed are extremely beneficial and provide more accurate levels of risk, the availability of the necessary data to obtain the required outcomes is often an issue (Bridges et al. 2003).

The use of integrated risk assessment procedures have become increasingly popular, with the need to assess both public health and environmental risks at the same time (Federa 1998, Jones et al. 2000, Bridges 2003). However, due to the fact that separate public health and environmental risk assessment processes have previously been used to assess these risks, variations in data type, amount and quality will vary significantly (Bridges 2003). Therefore, in most cases when undertaking an integrated approach, the need to identify and obtain data relating to all aspects (both public health and environmental risks) of the risk assessment process will be needed. The consequent use of this data will also entail similar approaches in modelling the inherent risks, which may be difficult due to the current processes and level of detail required to provide accurate quantitative assessments (Buchanan et al. 2000, Bridges
Therefore, the use of an Engineering risk assessment approach can be beneficial.

The engineering risk technique is formulated around the probability of failure models. That is, the level of risk associated with specific hazards is assessed through the probability of the hazard failing a specified target. The risk established through this process is equivalent to:

\[
Risk = probability\ of\ failure = P_f = P(L > R) = \int_{0}^{\infty} \int_{0}^{\infty} f_{LR}(L, R) dR dL
\]

(2.5)

where \( L \) = pollutant loading or concentration and \( R \) = resistance or prescribed water quality standard or threshold (Ganoulis 1996). In the case of integrated public health and environmental risk assessments, this basically assesses the probability of the level of a particular pollutant (for example nitrate or faecal coliforms) failing specified threshold levels, such as adopted water quality guidelines (ANZECC 2000 and NHMRC 1996). Utilising this technique, integrated risk assessment processes allows an acceptable means of assessing the different risk paradigms.

### 2.6.1.4 Risk Management and Mitigation

There are numerous risk management strategies presented in the literature (Chittaluru 1994, Fedra 1998, AS/NZS 4360:1999, CENR 1999, Chin and Hope and Peterson 2000, Brown and Root Services 2001, Lawrence et al 2001). These strategies are generally similar in what they want to achieve, but the processes involved in achieving it can be quite different. The reason for this is the dual definition of risk management. The definition of risk management describes the fundamental concept of controlling risk to provide minimal impacts arising from consequences. However, when describing the process of managing risk, the concepts of risk assessment and mitigation are also generally included. The basic concept of risk management is to generate viable options based on the assessment of the characterised risks. These options are then evaluated, with the inclusion of stakeholder perception, as this removes the decision maker’s perception of what is best to do (HB 203:2000). Inevitably, what is, and what is not acceptable will ultimately be a political and not a scientific issue (Fedra 1998). Whatever the case, the scientific background developed
for the overall risk assessment will greatly influence the ultimate decision as to what best management practice to adopt to mitigate the characterised risks to an acceptable level.

### 2.6.2 On-site Wastewater Treatment Management

The management of on-site wastewater treatment systems generally falls within the responsibility of the regulatory authorities. They are generally responsible for approving the installation of on-site systems, with approvals and restrictions based partly on their own regulations and the prescriptive criteria outlined in Australian Standards and codes. As a national standard, AS/NZS 1547:2000 should provide adequate OWTS assessment and management procedures to achieve the requisite sustainable outcomes nation-wide. However most regulatory authorities have in place regulations and guidelines for the use of on-site systems, which can alter substantially from one jurisdiction to another with regards to site assessment, system design, and more recently, management requirements. Unfortunately, this has been a major drawback to the nation-wide acceptance of AS/NZS1547:2000 and the adoption of standardised management strategies. As such, this has led to the realisation that more rigorous evaluations in regard to site assessments and the underlying ground conditions are needed. The prescriptive criteria used for the assessment of site and soil characteristics in on-site system design are not sufficiently robust enough to be used for all sites and all situations, which in turn have commonly led to poor system performance.

### 2.6.3 Risk-based Approach to On-site Systems

The evolving risk-based approach to on-site system siting, design and management can be considered as the next improvement to the current standards and codes for on-site wastewater treatment systems. It must be noted that risk has in general been implicitly included in current regulations for on-site wastewater treatment systems. These regulations include separation distances between soil adsorption fields and watertable levels, as well as setback limits between neighbouring properties and potable water supplies (Jones et al 2000). However, the implementation of these risk-
Chapter 2.0 On-site Wastewater Treatment

Based assessments have been modified to comply with the performance-based codes, therefore only aiding in the siting and design requirements rather than assessing the suitability for the system. In order to successfully implement risk assessment into regulatory codes, it is essential that it becomes the driving force behind the code, rather than being included as part of an existing code.

One of the most important issues specific to the development of a risked-based approach for on-site wastewater treatment is the inclusion of not only vast amounts of data necessary to perform specific risk assessments, but also the means of establishing a single coherent framework that includes both public health and environmental risks assessments, as well as means of assessing the risks associated with the siting, design and management aspects as well. This is probably the major limiting factor as to why a risk-based approach has not yet been developed in the regulatory procedures. Figure 2.15 provides a framework for risk assessment/management of wastewater soil absorption systems developed by Siegrist et al (2000). The most important challenge relating to the management of the inherent risks associated with on-site wastewater systems is the assessment of the magnitude of the risks in a given situation and the decision towards the most appropriate method of managing those risks (Siegrist et al 2000).

2.6.4 Overview of Current Risk Assessment/Management Models

As a result of the numerous reports of poorly performing OWTS and subsequent contamination of water resources, regulatory authorities worldwide are beginning to utilise risk-base models for assessing and managing the use OWTS. Under the SepticSafe™ program implemented in the State of New South Wales (NSW), Australia, following the Wallis Lake Hepatitis A outbreak, a risk-based model, the On-site Sewage Risk Assessment System (OSRAS) was developed (Brown and Root Services 2001, Kenway 2001). The OSRAS model was developed to utilise existing databases to determine the risk posed by on-site treatment systems on the surrounding environment and the cumulative impacts of systems on downstream environments.
Another similar model, the Development Assessment Module (DAM), has been developed by the Sydney Catchment Authority, NSW State, Australia (McGuinness and Martens 2003). DAM was developed in order to reduce the impacts on Sydney City’s water supply from new housing developments in areas relying on on-site systems for sewage treatment. DAM also utilises existing databases to predict the extent and direction of an effluent plume originating from a treatment system in order to assess its potential impact on water quality (McGuinness and Martens 2003). This predicted plume can then be utilised in assessments to determine the level of risk associated with installing an on-site system in a particular area.

Similar models have also been developed internationally, such as Trench™ 3.0 (Cromer 1999) which aids in the assessment and suitability of sites for septic absorption trenches; WARMF (Kirkland 2001), Watershed Analysis Risk Management Framework, which has been modified to incorporate effluent infiltration into the soil layer below the land surface to account for cumulative effects.

**Figure 2.15:** Conceptual framework for risk assessment/management of wastewater soil absorption systems. (Siegrist et al 2000)
of systems as non-point source pollutant loads (Chen et al. 2001); and MANAGE (Joubert et al. 1996) or Method for Assessment, Nutrient-loading, and Geographic Evaluation of Watersheds, a model used to identify groundwater pollution sources and future risks and to evaluate the impacts of alternative on-site wastewater treatment systems. However, though these models are able to determine the level of risk if an on-site system is installed at a particular location, the accuracy of the risk model is singularly dependent on the amount and type of data available to the user. Additionally, the complexity of some of these models reduces them to ‘black box’ approaches, whereby the user inputs data and receives an answer without any guidance as to how this was derived.

Another important limitation of these models is that they are unable to assess the level of risk currently present in an area due to existing on-site treatment systems. The cumulative impact approach, which assesses the impact on the environment and public health from incremental changes in the risk resulting from additional on-site systems, is assessed from a background or zero risk level. This may be misleading if the assessed area is already at high risk due to the presence of on-site treatment systems for many years.

For this reason, assessment and management frameworks have also been developed. Hoover (1998) developed a Framework for Site Evaluation, Design and Engineering of On-site Technologies Within a Management Context in order to addresses both risk management and long term operation and maintenance (Hover 1998). Similarly, NDWRCD (2002) utilised the framework developed by Hoover (1998) in order to develop a risk-based approach to OWTS assessment and management for the Tisbury region, Massachusetts, USA. Loetsher and Keller (2002) developed the SANEX system, a decision support tool created to allow regulatory authorities to assess the suitability of alternative OWTS systems for different sites. This framework utilises various site and soil characteristics, as well as alternative system technology to develop ranked indices to determine the most suitable system to use in a particular area. The main difference between utilising a developed risk model versus the development of a suitable framework for a particular region utilising OWTS is the incorporation of relevant stakeholders. This allows a more useful framework to be
devised based on the main issues associated for the area of concern, rather than trying to obtain and utilise the necessary required by developed risk models.

2.6.5 Summary of Key Research Literature Findings

It was readily observed throughout the literature that a number of deficiencies are evident in the currently adopted standards codes and guidelines employed for the assessment and management of OWTS. As a result of these deficiencies, the development of risk-based approaches for the assessment and management of OWTS are becoming increasingly popular as regulatory authorities identify the need to safeguard against environmental and public health issues. Several risk-based models developed for OWTS were reviewed through the literature. However, although these models provided a means of assessing the inherent risks, one of the major issues identified as lacking in both currently adopted standards and the risk-based models is the incorporation of scientifically defensible information for assessing the respective risks. Additionally, most of these developed models are integrated with sophisticated GIS packages, which can ultimately reduced them to ‘black box’ approaches with little flexibility allowed in the overall risk assessment process. Therefore, the development of a flexible risk-based framework that implements the necessary scientific information to assess site suitability and develop suitable management strategies to mitigate inherent risk needs to be developed.

2.7 Conclusions from Literature Review

On-site wastewater treatment systems are capable of providing adequate treatment and dispersal of domestic effluent provided they are situated in areas that retain appropriate site and soil characteristics and are managed and maintained on a regular basis. However, failure of OWTS is a common scenario, particularly for septic tank-soil absorption systems. This is related to several issues including unsatisfactory site and soil characteristics, general maintenance of systems, and the inappropriate assessment and management of on-site systems. Poor OWTS performance can lead to serious environmental and public health issues. Consequently, more appropriate assessment and management techniques need to be developed to safeguard against
the inherent hazards associated with the typical failure of OWTS. The evolving risk-based approach to on-site system siting, design and management can be considered as the next improvement to the current standards and codes employed in on-site wastewater treatment systems.

From the reviewed literature, several important issues were highlighted which needed to be investigated in order to develop a robust, scientifically based risk approach for the management of on-site wastewater treatment systems. These issues are:

- More thorough assessment of soil characteristics, based on scientific investigation and analysis, is required to ensure suitable effluent renovation is achieved through OWTS.
- Identification and assessment of the different factors that influence contaminant fate and transport of OWTS contaminants.
- Investigation of the overall extent of contamination of groundwater and surface water resources as a result of high densities of on-site systems.
- Assessment of faecal contamination of groundwater and surface water resources due to OWTS. This includes applying applicable means of identifying the respective sources of faecal contamination and linking human sources back to OWTS.
- Assessment of OWTS siting and design issues and the inherent environmental and public health risks.
- Development of a universal and scientifically robust integrated risk assessment framework for allowing more appropriate siting and assessment of OWTS to ensure adequate treatment performance is achieved.
- Development of a risk management process for mitigating the identified OWTS siting and design, environmental and public health risks.

The research conducted through this thesis explores these issues through a series of scientific papers, with the respective outcomes used for the development of an integrated risk framework for on-site wastewater treatment systems.
CHAPTER 3

FRAMEWORK FOR SOIL SUITABILITY EVALUATION FOR SEWAGE EFFLUENT RENOVATION

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This Chapter is an exact copy of the journal paper referred to above.
CHAPTER 4.0

ASSESSMENT VIA DISCRIMINANT ANALYSIS OF SOIL SUITABILITY FOR EFFLUENT RENOVATION USING UNDISTURBED SOIL COLUMNS

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This Chapter is an exact copy of the journal paper referred to above.
This paper is not available online. Please consult the hardcopy thesis available from the QUT Library.
CHAPTER 5.0

USE OF CHEMOMETRICS METHODS AND MULTICRITERIA DECISION-MAKING FOR SITE SELECTION FOR SUSTAINABLE ON-SITE SEWAGE EFFLUENT DISPOSAL

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This Chapter is an exact copy of the journal paper referred to above.
Chapter 5.0 Use of chemometrics methods and multicriteria decision-making for site selection for sustainable on-site sewage effluent disposal


This paper is not available online. Please consult the hardcopy thesis available from the QUT Library
CHAPTER 6.0

ASSESSMENT OF HIGH DENSITY OF ONSITE WASTEWATER TREATMENT SYSTEMS ON A SHALLOW GROUNDWATER COASTAL AQUIFER USING PCA

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Supervised and assisted with manuscript compilation, editing and co-author of manuscript.

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Chapter 6.0 Assessment of High Density of Onsite Wastewater Treatment Systems on a Shallow Groundwater Coastal Aquifer using PCA

*Environmetrics (Article in Press - DOI: 10.1002/env.698)*

This paper is not available online. Please consult the hardcopy thesis available from the QUT Library
CHAPTER 7.0

ENVIRONMENTAL AND ANTHROPOGENIC FACTORS AFFECTING FECAL COLIFORMS AND *E. coli* IN GROUND AND SURFACE WATERS IN A COASTAL ENVIRONMENT

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Chapter 7.0 Environmental and anthropogenic factors affecting fecal coliforms and *E. coli* in ground and surface waters in a coastal environment

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CHAPTER 8.0

SOURCING FECAL POLLUTION FROM ONSITE WASTEWATER TREATMENT SYSTEMS IN SURFACE WATERS USING ANTIBIOTIC RESISTANCE ANALYSIS

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Supervised and assisted with manuscript compilation, editing and co-author of manuscript.

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CHAPTER 9.0

INTEGRATED RISK FRAMEWORK FOR ONSITE WASTEWATER TREATMENT SYSTEMS

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This paper is not available online. Please consult the hardcopy thesis available from the QUT Library
Rapid development around the urban fringes and hinterland regions in Southeast Queensland has dramatically increased the need for on-site wastewater treatment systems (OWTS) for the treatment and dispersal of domestic wastewater. Due to the absence of centralised treatment facilities, OWTS are the most economical and feasible alternative for treatment of wastewater. However, on-site system failure is a common issue both in Australia and internationally (Hagedorn et al 1991, Geary 1992, Hoxley and Dudding et al 1994, Harris 1995, Goonetilleke et al 2000, Goonetilleke et al 2002). Typically, poor treatment performance of OWTS is a result of several factors including inappropriate soil and site characteristics, poor operation and maintenance practices and a general lack of knowledge by householders regarding appropriate use and general maintenance of OWTS (Whitehead and Geary 2000, Alle et al 2001). The current standards and guidelines used for the assessment and management of OWTS, although performance based, have shown to be inadequate for preventing system failure, leading to environmental and public health impacts (Whitehead and Geary 2000, Seigrist 2001, Dawes and Goonetilleke 2004). The assessment and management of OWTS generally falls within the responsibility of individual local authorities. Consequently, wide variations in adopted codes and guidelines between regulatory authorities are common. As such, the need to develop universal and scientifically robust OWTS management guidelines is necessary (Beavers 1999). Risk-based approaches to OWTS assessment and management are considered one of the most appropriate and functional means of achieving this objective. However, risk-based concepts for OWTS have not been widely implemented, mostly due to a lack of knowledge and guidance on the practical implementation of risk-based processes (EPRI 2001).

The research conducted contributed to overcoming the constraints associated with OWTS siting, design and management, including more thorough soil suitability assessment techniques, assessment of the inherent factors associated with the fate
and transport of contaminants from OWTS, and the integration of this research into the development of the risk-based framework. The research outcomes achieved were applied to the development of an integrated risk-based approach for the management of OWTS. The primary objectives for the research project were achieved by addressing the specific research aims, which included investigations into soil suitability for effluent renovation, understanding contamination and transport processes influencing environmental and public health risks and the development of an integrated risk framework for OWTS. Several complementary studies were undertaken focusing towards achieving these specific aims, with the research conducted and relevant outcomes discussed in the enclosed scientific papers in Chapters 3 to 9.

### 10.1 Soil suitability for effluent renovation

The suitability of soil plays a prominent role in providing adequate treatment performance and dispersal of effluent from an OWTS. For common septic tank-soil absorption systems, the soil surrounding the dispersal area is required to provide adequate attenuation and removal of effluent pollutants, such as nutrients and pathogens, as well as providing an effective means to disperse the discharged effluent (Dawes and Goonetilleke 2004). The soil also provides a treatment medium for aerobic wastewater treatment systems (AWTS) employing surface irrigation. Although most of the effluent discharged to the surface application area will be utilised by vegetation or transmitted to the atmosphere via evapotranspiration, the pollutants themselves will remain behind in the soil. Therefore, the ability of the soil to attenuate and remove these pollutants needs to be assessed. However, the current methods of assessing soil suitability for use in OWTS is typically focused on the dispersal of effluent, with little attention given to whether the soil is capable of providing adequate treatment. In recent times however, the need to pay more attention towards soil treatment of effluent has been recognised due to the high failure rates of OWTS reported both nationally and internationally (Wells 1989, Geary 1992, Whitehead and Geary 2000, Carroll et al 2004, Dawes and Goonetilleke 2004).
The main deficiency in the current means of assessing soil suitability for OWTS is the lack of scientific methodology for identifying which soils are suitable for providing adequate renovation of discharged effluent. The developed soil suitability framework (Carroll et al 2004; Chapter 3) addresses some of these deficiencies by assessing the renovation suitability of a number different soil types common throughout Southeast Queensland. This was based on three major mechanisms, soil renovation ability, soil permeability and soil drainage. The procedure used for determining this is shown in Figure 10.1. Soil renovation ability was evaluated by analysing a number of soil physico-chemical characteristics employing multivariate statistical methods such as PCA and multicriteria decision making methods such as PROMETHEE and GAIA. The analysed soils were ranked based on their effluent renovation ability. The main focus of the soil research undertaken was to determine which of the major physico-chemical characteristics influenced soil renovation ability, and thereby prioritise the different soil types based on their overall suitability.

Detailed descriptions of the analytical processes and multivariate assessment techniques used for assessing the suitability of soils are described in Carroll et al (2004a,b; Papers 1 and 2) and Khalil et al (2004; Paper 3). The multivariate analysis undertaken on the soil variables showed that there are several primary variables that influence the ability of soil to renovate effluent. These include the amount and type of clay present in the soil, Cation Exchange Capacity (CEC), organic matter content (%OM) as well as soil permeability and drainage characteristics. CEC defines the ability of a soil for attenuating and removing effluent pollutants, and can be significantly influenced by the amount and type of clay present as well as the %OM. Therefore, more suitable soils would retain higher CEC values. However, both the permeability and drainage can also be influenced by the amount and type of clay present in the soil. Therefore lower amounts of clay are more appropriate. For a soil to provide suitable treatment and dispersal of effluent, it must fulfil both the treatment and dispersal processes.
Figure 10.1: Soil renovation suitability framework (adapted from Carroll et al 2004)

From the outcomes of this study, prioritisation of the respective soils was conducted.

Table 10.1 provides the obtained suitability rankings for the investigated soil types. The major soil groups that were identified as most suitable for effluent renovation include the Chromosol, Kurosol, Ferrosol and Dermosol soil groups. These results are significant as soils previously thought to be incapable of providing appropriate treatment of sewage effluent based purely on soil physical characteristics were subsequently found to be suitable. The most important outcome from the soil investigations was in relation to Kurosol soils. Kurosols which were previously defined as inappropriate for effluent application based purely on drainage characteristics (Noble 1996), was found to be one of the most suitable soils within the Gold Coast region for providing effluent renovation. However, this particular finding signifies the necessity for identifying suitable soils based on the combination of renovation ability, permeability and drainage characteristics.

<table>
<thead>
<tr>
<th>Soil Classification</th>
<th>Suitability</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chromosols, Kurosols</td>
<td>1 – Very Good</td>
</tr>
<tr>
<td>Ferrosols, Dermosols</td>
<td>2</td>
</tr>
<tr>
<td>Kandosols, Rudosols</td>
<td>3</td>
</tr>
<tr>
<td>Podosols, Tenosols</td>
<td>4</td>
</tr>
<tr>
<td>Organosols</td>
<td>5</td>
</tr>
<tr>
<td>Vertosols, Sodosols</td>
<td>6</td>
</tr>
<tr>
<td>Hydrosols</td>
<td>7 – Very Poor</td>
</tr>
</tbody>
</table>
Similar results to those derived by Carroll et al. (2004; *Paper 1*) were also developed by Khalil et al. (2004; *Paper 3*) in a study undertaken to assess land capability for on-site sewage effluent renovation in the Logan City Council region. Soils with a higher ability for attenuating and removing pollutants were determined based on CEC and %OM in soil.

To further clarify the soil suitability rankings obtained from the previous study (*Chapter 3*), the effluent renovation ability of several of the most common soil types found in Southeast Queensland, including the Gold Coast, was investigated using undisturbed soil columns (Carroll et al 2004; *Paper 2*). The use of soil columns for assessing different soil types for treating effluent have been investigated in numerous past research studies (Parker and Carbon 1981, Powelson and Gerba 1994, Jin 2000, Van Cuyk et al 2001). However, most of these studies utilised packed soil columns (homogeneous soil repacked into columns). Although indicating the capacity of the soil medium to remove contaminants, the results can be misleading as the utilised medium is not equivalent to in-situ conditions. Additionally, other studies used soil columns for investigating specific processes. For example, soil columns have been used for assessing pollutant attenuation and removal characteristics with little regard to the effluent dispersal process of the soil (Lam et al 1993, Jin et al 2000). Additionally, several studies have used soil columns for assessing long term effluent acceptance rates through the development of a biological clogging layer (Laak, 1970, Daniel and Bouma 1974, Kristiansen 1981, Siegrist 1987). However, investigation into the treatment and dispersal mechanisms of the soil medium itself was not investigated.

The purpose of this column study (*Chapter 4*) was initially to assess the suitability of different soil types for effluent renovation, including contaminant attenuation and removal, as well as the soil dispersal processes. Additionally, the respective changes occurring to the soil medium as a result of effluent application were also investigated. This included contaminant wash-off processes, soil degradation and permeability and drainage processes. The long-term behaviour of the soil cores under effluent application was also investigated, with particular reference to the renovation and dispersal characteristics simultaneously with changes in the soil’s textural and physico-chemical makeup. This multifaceted approach has not been conducted in
other soil column studies published to-date, particularly in relation to effluent renovation processes. The results obtained from this column study after eight months of effluent application agreed with the previous prioritised soil rankings as discussed by Carroll et al (2004; Paper 1). Soils that retained an initially high CEC, %OM, medium clay content, moderate permeability and good drainage were found to be more suitable for the long-term renovation of effluent. However, several soil columns that had very high clay content did not adequately transmit the applied effluent to allow long term use. Comparatively, it was observed that soils with a high permeability, and low CEC, %OM and %C did not provide sufficient attenuation and removal of effluent pollutants. Hence, these soil types are not suitable for the treatment and dispersal of effluent from OWTS.

Although the focus of the research project was the Gold Coast region, the outcomes of these studies are generic and are of significance to the Southeast Queensland region in general. The soil types investigated are common to the region. Therefore, the results and established soil suitability framework will be beneficial to other regions employing OWTS. Adoption of the developed framework in addition to site assessment and management techniques will provide more appropriate identification of problem soils and minimise the high number of poorly performing OWTS due to inappropriate soil conditions. This will ultimately minimise the inherent environmental and public health risks as a result of the poor system performance.

10.2 Contamination and transport processes influencing environmental and public health risks

Contamination of water resources as a result of poor treatment performance of OWTS can have serious environmental and public health impacts. Increased nutrient loads into surface waters can cause nutrient enrichment, resulting in eutrophication and algae blooms. Apart from the environmental degradation of water resources and the resulting aesthetic water quality issues, this can also cause public health hazards. High levels of nitrate in drinking water sources are regarded as a public health issue, causing health related issues such as methaemoglobinemia (Blue Baby Syndrome), development of carcinogenic nitrosamines (cancerous cells developed through
bacterial production \textit{N-nitroso} compounds), increased infant mortality, and changes to the immune system (Bouwer and Idelovitch 1987, Fewtrell 2004). Additionally, faecal contamination as a result of inadequately treated effluent transported to water resources is also of critical concern to public health. Pathogenic organisms are present in inadequately treated wastewater. The inherent possibility of disease resulting from contact with pathogenic organisms in contaminated water is high. These issues are further exacerbated by high densities of OWTS situated in areas without adequate soil and site characteristics available to provide suitable effluent treatment.

The movement of pollutants from OWTS into the groundwater, and eventually surface water are reliant on a number of physical and chemical processes which may either hinder or advance contaminant transport. The most critical factor that influences these mechanisms is the soil. However, some soils, particularly sandy soils or soil influenced by seasonal or permanent saturation, do not provide adequate renovation ability. Numerous studies have shown that the siting of OWTS on these soils types increase the risk of contamination of both groundwater and surface water, particularly in areas with high OWTS densities (Perkins 1984, Sinton 1986, Geary and Whitehead 2001, Carroll and Goonetilleke 2004). Two studies conducted through this research; Carroll and Goonetilleke (2004; Paper 4) and Carroll et al (2004; Appendix A), also indicated substantial contamination of water resources as a result of high densities of OWTS. In the course of the research undertaken, the impact of high densities of OWTS on a shallow groundwater aquifer was assessed using multivariate analysis. Groundwater samples collected over a four month period were assessed for nutrient and microbiological contamination. These studies indicated that significant nitrate and faecal contamination of shallow groundwater aquifers were evident in urban developments utilising OWTS for treatment and dispersal of effluent. The high correlation found between contaminants typical of effluent indicates that the major source of the contamination results from OWTS. Additionally, the influence of high densities of OWTS also impacted on the adjacent estuarine surface waters, with both faecal contamination and elevated nitrate and phosphate concentrations above stipulated water quality guidelines (ANZECC 2000).
An additional issue that needs to be adequately recognised in assessing and managing OWTS for environmental and public health protection is the implementation of suitable setback distances between OWTS and water resources. Current setback distances employed for addressing environmental and public health safeguards can be inadequate where soils and shallow groundwater aquifers are concerned, as highlighted in several past studies (Cromer et al 2001, Pang et al 2003). Although not investigated as part of this research, results of past studies undertaken have indicated that the transport of chemical compounds, particularly nitrates, can be far greater than the adopted setback criteria. However, groundwater and surface water investigations conducted through this research (data not published) found that the use of OWTS in areas that had suitable soil for effluent renovation did not cause detrimental impacts on water resources.

Similarly microbiological contamination of water resources is a critical concern. This is of particular importance with regards to the appropriate assessment and management of OWTS, as poorly performing systems can result in adverse public health impacts. Numerous cases of viral and bacteriological infections from the contamination water sources as a result of failing OWTS have been reported in the literature (Fliesher et al 1998, Cliver 2000, Borchardt et al 2003). Although major disease outbreaks are generally reported through the literature for their public health significance, the overall extent of public health issues associated with failing OWTS may be far more substantial. The significance of infection is commonly considered minor compared to these larger outbreaks. A recent study undertaken by Fleisher et al (1998) investigating the risk of illness associated with bathing in waters contaminated by domestic sewage suggests that these relatively minor incidences can be a significant public health issue. Microbiological contamination of bathing water was found to increase the risk of contracting gastroenteritis, acute febrile respiratory illness, ear and eye infections.

The most common method for assessing water quality in relation to public health and the potential for the presence of pathogenic organisms is through the use of faecal indicators, commonly faecal coliforms and *Escherichia coli* (*E. coli*). Although pathogenic organism themselves may be able to provide more appropriate information regarding the risk to public health, the time and costs associated with the
enumeration of these organisms, particularly the pathogenic organisms of concern, are far in excess compared to enumeration of faecal coliforms. Consequently, most authorities utilise faecal indicators to assess the microbiological water quality in relation to faecal contamination and resulting pathogenic organisms. This leads to several significant issues which need to be adequately considered when assessing the risk to public health from OWTS. Faecal bacteria can be emitted from various sources, including agricultural sources, wild and domesticated animals, urban development and effluent treatment facilities such as OWTS (Kelsey et al 2004). Although confirming that faecal contamination is apparent, indicators may not accurately portray the transportation and survival of other pathogenic organisms they are intended to show. This is compounded by the fact that the faecal indicators may not be from one particular source, but rather from a variety of sources in the region. However, the presence of faecal bacteria in water resources does indicate that faecal contamination has occurred (Meays et al 2004).

It is generally considered that fluctuations in faecal bacteria numbers in water resources are influenced by rainfall, with higher levels occurring after high rainfall periods (Kelsey et al 2004, Ackerman 2003, Muirhead 2004). This phenomenon was also observed through the assessment of faecal contamination through two studies undertaken in the course of this research project (Carroll and Goonetilleke 2005; Paper 4 and Carroll et al 2004; Paper 5). Increases in faecal contamination of water resources, particular surface water, occur after large rainfall events due to the resulting runoff collecting faecal bacteria from numerous sources within a catchment. However, in studies undertaken by Lipp et al (2001) and Alm et al (2003), high numbers of faecal bacteria were also observed irrespective of rainfall characteristics. Therefore, research was conducted to investigate whether other environmental and anthropogenic factors also influence the contamination and transport processes of faecal bacteria (Carroll et al 2004; Paper 5).

Several past research investigating the various factors influencing the transportation of faecal bacteria have found that increased urbanisation, combined with rainfall significantly increase the extent of faecal contamination (Young and Thackston 1999, Kelsey et al 2004). However, limited studies have specifically investigated the different factors than can influence the transportation of faecal bacteria associated
with OWTS treatment and dispersal. This was the main focus of the conducted study. Several factors, including rainfall, OWTS density, urbanisation, and soil type, were found to enhance the transportation of faecal bacteria through the environment.

However, the factors influencing transportation were found to differ between surface and groundwater. Surface water contamination was significantly influenced by rainfall in association with increased urbanisation, mostly due to the higher percentage of impervious areas associated with urban environments. This results in higher runoff causing substantial increases in faecal bacteria loadings to surface water. Only minor correlations with the remaining factors, particularly OWTS density and slope, were observed. Comparatively, the factors that greatly affected faecal contamination of groundwater included higher densities of OWTS, soil type and resulting depth to ground water, with only a minor influence exerted by rainfall. Additionally, through this research it was found that the combined effect of multiple factors significantly affected faecal bacteria loadings more than any individual factor alone. Therefore, the assessment of faecal contamination for water resources should be conducted as a multivariate study to investigate the significance of all related factors. Although certain individual factors may indicate a stronger influence, the cumulative effect caused by several minor factors could be more significant in influencing the transport of faecal bacteria to surface waters.

Additionally, an important feature of this research and generally not undertaken in most water quality assessments until recently, was the identification of the respective sources of faecal bacteria using Bacterial Source Tracking (BST) methods. Faecal coliform bacteria inhabit the intestinal tract of all warm-blooded animals and consequently, faecal coliform counts from a contaminated waterway will not provide any information as to the actual source of contamination. This information is important as faecal pollution resulting from human sources will establish a high public health risk due to the possible presence of pathogenic organisms. Additionally, if the faecal source is known, suitable management actions can be implemented to prevent further contamination and to mitigate the health risks (Harwood et al 2000).
In order to perform a more appropriate public health risk assessment, within the BST method of analysis, Antibiotic Resistance Patterns (ARP) of enumerated *E. coli* isolates was undertaken (Carroll et al 2004; *Paper 6*). The major advantages of utilising ARP techniques over molecular methods such as random amplified polymorphic DNA or rep-PCR DNA extraction methods (Pareveen et al 1999, Dombeck et al 2000), is that ARP profiles can be used on more inclusive taxonomic groups of faecal coliforms and faecal streptococci, with hundreds of faecal isolates able to be analysed within a few days of sample collection, at a fraction of the cost of molecular methods (Whitlock et al 2002). ARP essentially utilises the resistance of selected faecal bacteria isolates. For this study, the resistance of *Escherichia coli* (*E. coli*) to several antibiotics at varying concentrations were investigated in order to obtain their resistance profiles. The underlying assumption of the ARP technique is that due to the increased use of antibiotics by humans and domesticated animals, isolated *E. coli* bacteria from these host sources will have higher resistance than that of wild animals (Wiggins 1996). The ARP technique required a library of known *E. coli* isolates, from human and non-human sources, to be tested for their respective ARP. These were then analysed statistically using multivariate discriminant techniques to separate the respective patterns into source groups. Once the known source library was developed, *E. coli* from the investigated water samples were tested for their ARP and compared to the known source library and categorised according to the respective grouping of known source isolates with similar ARPs.

Previous studies have demonstrated the successful application of ARP for discriminating sources of faecal contamination in water sources (Wiggins 1996, Wiggins et al 1999, Booth 2003). However, although indicating that faecal bacteria from human sources can be significant, no linkage towards the actual origin of this contamination has been suggested. This was an additional aim of the source tracking study undertaken in this research project. With the only available source of human faecal bacteria being from OWTS within the investigated catchments, *E. coli* isolates categorised as human could only have originated from these systems. The procedures used in developing the source library and determining the respective sources for the two monitored catchments is discussed in Carroll et al (2004; *Paper 6*).
The main outcome from this study indicated that higher percentages of human *E. coli* in collected water samples were in those areas that did not have appropriate soil and site conditions for sewage effluent treatment. This is in agreement with the earlier study (*Chapter 7*), which showed that concentrations of faecal coliforms and *E. coli* were influenced by the type of soil in the monitored catchments. However, of more significance and apparent throughout all monitored locations, was that during drier periods, higher percentages of human *E. coli* isolates were found in water samples when compared to non-human source isolates. This is related to two significant factors. Firstly, rainfall resulted in an increase in the overall number of faecal bacteria in the water source. This was due to increased numbers of faecal bacteria being washed into the surface water with surface runoff from the different sources, such as domesticated animals and wildlife located throughout the catchments. Consequently, higher percentages of non-human sources were more dominant during these high rainfall periods. However, during drier conditions, the main source of contamination was found to be of human origin, particularly as the investigated waterways flowed through the urbanised developments using OWTS. Therefore, although the overall magnitude of faecal bacteria numbers increases substantially after rainfall, the respective sources of contamination are more evenly distributed between several sources. However, during dry conditions, although lower overall numbers of faecal bacteria were observed, the major source was found to be of human origin. This suggests that a continuous source of human faecal contamination, such as a result of poorly performing OWTS, is present. This is an important issue that needs to be considered when assessing the public health risks to water resources. The likelihood of human contamination from OWTS is more apparent during drier conditions.

### 10.3 Integrated Risk Framework for OWTS

The utilisation of risk-based approaches for OWTS has become more prominent in recent years due to the inherent environmental and public health concerns caused by poor OWTS performance. Although several risk-based approaches have been developed for the assessment and management of OWTS, several deficiencies in these approaches have however undermined their overall usefulness and acceptance. These include:
1. lack of scientific data and knowledge on the development of the risk process and assessment of risks associated with the failure of OWTS;
2. difficulty in the development of a framework for the incorporation of OWTS siting and design, environmental and public health risk assessment into a single integrated approach;
3. common approach of assessment of risks based on computer models rather than real world scenarios, which in turn requires subjective user input and interpretation;
4. no assessment of existing environmental and public health risk levels associated with OWTS is undertaken. Modelled risk scenarios are generally based on cumulative risk processes established from baseline or zero risk levels; and
5. lack of assessment of risk associated with groundwater contamination caused by OWTS.

In developing a scientifically robust and cohesive risk-based framework that is generic and can be universally applied, these deficiencies need to be investigated. This will allow the integration of scientific knowledge into risk assessment and management concepts, with the implementation of appropriate performance goals for successful management and mitigation of the associated risks.

The main aim of the current research project was to satisfy these identified research deficiencies and to utilise the respective outcomes in the development of a risk-based approach to OWTS siting, design and management. Consequently, an integrated risk framework was developed to accommodate this risk-based approach. Figure 10.2 provides the generic risk framework developed. The respective processes utilised and risk assessment developed through the construction of this framework is described by Carroll et al (2004; Paper 7).
The integrated risk assessment framework for OWTS has several innovations which are typically not undertaken in existing approaches. The developed risk assessment process adopts an integrated approach, with the aim of providing an assessment of the different risk facets into a single cohesive process. This eliminates the issue of identifying and characterising the risks associated with OWTS, with separate assessments undertaken for the environmental and public health risks. Individual risk assessment approaches tend to be approached using different methodologies. This inadvertently causes different variations in risk assessment processes, techniques and data utilisation to be adopted and therefore introduces uncertainty throughout the assessment process. Integration of the different risk facets removes these problems (Bridges 2003, Federa 1998). Additionally, the developed framework quantifies the respective levels of risk associated with OWTS based on established scientific data analysis, which has not been adequately implemented into other OWTS risk assessment approaches.

The assessment of risks undertaken in existing risk models such as OSRAS (Kenway et al 2001), are based on available data which in most cases is derived from anecdotal or empirical relationships and may not be scientifically defensible. Additionally, risk associated with public health and environmental hazards are...
obtained from model outputs, which can produce uncertainty throughout the risk assessment phase. Thirdly, the assessment of the inherent risks associated with groundwater as a result of OWTS has not been utilised in any existing risk assessment process developed to-date in Australia, known to the author. The main reason for this is that most local regulatory authorities in Australia, particularly in Southeast Queensland, do not have sufficient information on groundwater conditions within their area. This is generally related to the fact that most existing water supplies are obtained from surface water resources. However, the need to protect groundwater resources is becoming increasingly important. Due to the increasing need to identify potential new water sources for rapidly developing areas, groundwater will become important in the future.

The developed integrated risk framework consists of three major stages for the assessment and management of the risks associated with OWTS. The integrated risk assessment process established through Stage 1 (Risk assessment and risk map development) allowed the assessment of the risks related to OWTS siting and design (this involves an assessment of the contributing hazards related to site and soil characteristics, landscape positioning and planning issues), environmental and public health hazards to be undertaken. Quantification of these risks was conducted through field investigations and analysis of scientific data, as described in Chapters 3 to 8. Once quantified, the characterised risks associated with each identified hazard were incorporated into a GIS and risk zone maps were produced to spatially display the characterised risks. The developed GIS database combined the individual risk layers into a single risk map indicating low and ‘at risk’ (medium and high risk) areas via a boolean overlay process.

Stage 2 (Detailed assessment) of the risk framework details the level of assessment and management required to ensure proper treatment performance of OWTS is achieved. Areas indicated as being ‘at risk’ were required to undergo further detailed assessment to ensure that the most suitable treatment system which will not lead to adverse environmental and public health impacts is utilised. Due to the framework’s flexibility, the exact assessment requirements to be utilised is at the discretion of the stakeholders and regulatory authorities. However, as an example, for areas indicated as being ‘at risk’ due to poor soil conditions, more detailed soil investigation,
including scientific analysis of soil samples should be conducted to assess the soil suitability of the investigated area. Additionally, more thorough investigations of the other associated hazards contributing to ‘at risk’ areas should be conducted.

Finally, Stage 3 (Management and mitigation) involved employing suitable management and mitigation measures to ensure that the characterised risks were suitably managed. Similar to Stage 2, the implementation of appropriate management protocols for mitigating assessed risks is at the discretion of the stakeholders and local authorities. This was due to the fact that differences in the limiting hazards, and appropriate management protocols may differ between local authorities. Consequently, to ensure that the developed framework was sufficiently flexible to be universally accepted, the implementation of specific management processes at a generic level was not appropriate. Appropriate risk management processes should be developed to specifically target the identified risks, and not developed as standard techniques.
CHAPTER 11.0
IMPLEMENTATION OF INTEGRATED RISK FRAMEWORK,
GOLD COAST REGION

The research undertaken for this project was conducted in collaboration with Gold Coast City Council (GCCC). The generic integrated risk framework developed was modified with the implementation of the necessary assessment and management requirements to allow incorporation into the Gold Coast City Planning Scheme. The modified framework developed for GCCC is given in Figure 11.1. Two major modifications were incorporated into this framework, including the additional processes developed for the improved assessment of ‘at risk’ regions and the inclusion of a critical point monitoring (CPM) program as an additional management strategy. The implementation of the CPM program is discussed in detail in Section 11.5. The respective processes for developing and implementing the integrated risk framework for assessment and management of OWTS for Gold Coast City Council are discussed below.

11.1 Problem Formulation and Hazard Identification

Following the general risk assessment/management framework as specified in AS 4630:1999, the first stage in implementing the OWTS risk framework was problem formulation and identification of the respective research tasks to be undertaken. This entailed the following; (i) identification of relevant stakeholders who will need to participate in the project, (ii) identification of areas within Gold Coast region that are highly sensitive to the use of OWTS, (iii) development of a logical process for progressing from initial project development to final framework implementation, and (iv) identification of the important hazards associated with OWTS that are critical for the Gold Coast region. These issues were resolved by conducting several stakeholder workshops held in conjunction with Gold Coast City Council.
CHAPTER 11.0 Implementation of Integrated Risk Framework, Gold Coast City

Integrated Risk Assessment Process

Environ. Risk Assessment
- Assess Nutrient Concentration
  - Phosphorus
  - Nitrogen
Risk = probability of failure = P(L > R)
L = Pollutant conc
R = Water quality threshold

PH Risk Assessment
- Assess Microbiology
  - Faecal Bacteria
  - BST - source
Risk = probability of failure = P(L > R)
L = E.coli conc x "human classification"
R = Water quality threshold

OWTS siting and design
- Soil
- Lot Size
- Setback Distances
  - Slope
  - Flood Plain

Risk = probability of failure = P(L > R)
L = Pollutant conc
R = Water quality threshold

Assessment of slope based on current guidelines and literature. Values differ regarding surface or subsurface dispersal.

Assessment of OWTS in High Risk areas
- Soil
- Slope
- Flood Plain

Additional soil tests and assessment based on Soil suitability framework (Carroll et al 2004) to determine level of risk.

Assessment of OWTS in High Risk areas
- Amend Existing
- New System
- Density
- Lot Size (Planning)
- Setback distances on lot
- Setback distances off lot

Can new system be installed with suitable setback distances while minimising risks.

Assessment of risk involving system density after system amendment.

Will new system increase density adversely affecting assess risks.

Risk Management and Mitigation Protocols

1. Needs to be on A3
2. figure title and number?
All stakeholders who were perceived to have an interest in the development and implementation of the integrated risk framework were identified. These stakeholders included Gold Coast City Council officers, OWTS regulators, environmental health officers, developers, plumbers and inspectors, soil assessors, risk assessors and community groups. The inclusion of stakeholders was fundamental for the development of the framework with their involvement throughout the decision making process. Additionally, with the current lack of scientific information and knowledge in the use of OWTS, the inclusion of stakeholders throughout the development of the framework allowed the transfer of knowledge and scientific advances achieved in the assessment of OWTS. The initial identification of sensitive areas in relation to OWTS in the Gold Coast region was undertaken in discussions with the stakeholders. Several areas were identified as highly sensitive due to both, environmental and public health issues. These areas are shown in Figure 11.2. The identification of sensitive areas was achieved through the assessment of several parameters including unsatisfactory soil and site characteristics, contamination of water resources, failing systems or high densities of OWTS. These identified areas formed the basis for several field investigations and assessments as discussed in Chapters 3-9.

The next stage in the project was to develop a logical progression from the identification of sensitive areas through to the processes of hazard identification, risk assessment and management, framework development and finally the implementation of the framework into the GCCC Planning Scheme. Based on the information received from the various stakeholders, a Logic Model as described by McLaughlin and Jordan (1999) was developed as shown in Figure 11.3. The model outlines the use of logical sequences to identify the required outcomes from the project and the steps and processes to be followed to successfully to achieve these outcomes. This allowed the stakeholders to visualise how the project was to be formulated and developed, through to the conducting of field investigations and the analysis of collected data. It also allowed the identification of various stages at which the communication and transfer of scientific information and outcomes back to the stakeholders was crucial.
Figure 11.2: Areas within Gold Coast Region initially identified as sensitive to OWTS and resultant environmental and public health issues
**Figure 11.3**: Logic model showing the logical sequences from project formulation to implementation of risk framework for Gold Coast City.
The final stage in the formulation of the project was to identify the key OWTS hazards that were significant for the Gold Coast region. Although all inherent hazards need to be identified and taken into consideration, the identification of hazards that are the most significant allowed the focus of the research to be conducted towards the main concerns in the research area. The key hazards identified through this process are described in Table 11.1. From the identification of the hazards involving OWTS siting and design, public health and environmental impacts, several initial criteria, including soil suitability and planning and setback distances that reflected these hazards were selected. A preliminary risk assessment was undertaken to develop an initial risk map for the Gold Coast region. These criteria and the resulting map are provided in Table 11.2 and Figure 11.4 respectively. The initial risk map enabled the identification of other potentially sensitive areas in addition to those already identified. These additional areas were also investigated as part of the research program undertaken. Through the logical development of the risk framework, the initial risk map was refined regularly as the research progressed and additional scientific information became available.

Table 11.1: Key Hazards and contributing factors related to OWTS

<table>
<thead>
<tr>
<th>Item</th>
<th>Key Hazard</th>
<th>Contributing Factors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Public Health</td>
<td>Contamination of water/surrounding environment such that a considerable health risk is evident due to the release of contaminants (namely pathogens) which have an impact on human health</td>
<td>9. Surface exposure 10. Water supply (ground/surface) 11. Aerosols 12. Pests (mosquitoes etc)</td>
</tr>
<tr>
<td>Environmental</td>
<td>Release of contaminants into the receiving environment (ground/surface waters) causing environmental degradation such as eutrophication.</td>
<td>9. Surface runoff 10. Groundwater discharge 11. Flooding 12. Water table</td>
</tr>
</tbody>
</table>
Table 11.2: Criteria used for establishing preliminary risk maps

<table>
<thead>
<tr>
<th>Soil Criteria</th>
<th>Risk</th>
<th>Criteria</th>
<th>Implication</th>
</tr>
</thead>
</table>
|               | High | • Soils that have imperfect or poor drainage ability  
• Hydrosol Soils; soils that are seasonally or permanently saturated | • Soils that have poor drainage inhibit the disposal of effluent through the soil, which reduces the soils renovation ability.  
• Hydrosol soils, although generally well drained sandy soils, are saturated, making drainage poor. |
|               | Medium | • Soils that are moderately well drained  
• Anthroposols (man-made soils) and soils which have been altered | • Moderately well drained soils allow slow drainage, which can affect the soils renovation ability |
|               | Low | • Soils that are well drained | • Soils that have good drainage, increase its ability to renovate effluent |

<table>
<thead>
<tr>
<th>Planning Criteria</th>
<th>Risk</th>
<th>Criteria</th>
<th>Implication</th>
</tr>
</thead>
</table>
|                   | High | • Less than 0.4 Ha  
• Urban residential areas developed for high-density housing which are provided with reticulated water, but utilise on-site wastewater systems | Minimum lot size for developments in these residential areas must not be less than:  
1. Residential - 400m²  
2. Detached dwellings-600- 2000m²  
3. Village -600m²  
4. Hinterland subdivision -4000m² |
|                   | Medium | • 0.4 to 4 Ha  
• Park Living residential areas developed for low-density housing with reticulated water and utilise on-site wastewater treatment | Lot sizes must not be less than 8000m² minimum and no larger than 4 Ha |
|                   | Low | • Greater than 4 Ha  
• Rural residential areas utilising both on-site wastewater treatment and water supplies | Rural residential areas with lot sizes greater than 4 Ha, with maximum lot sizes up to 20 Ha |
|                   | Sewered | Urban residential areas with high density housing with both reticulated water and sewerage | Research not required in this area. |

<table>
<thead>
<tr>
<th>Environmental Sensitivity Criteria</th>
<th>Risk</th>
<th>Criteria</th>
<th>Implication</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>High</td>
<td>Less than 100m from nearest water source</td>
<td>Greater risk of contamination of surface water resources from both surface and subsurface flow.</td>
</tr>
<tr>
<td></td>
<td>Medium</td>
<td>Between 100 and 500m from nearest water source</td>
<td>May impose some risk of contamination from surface and subsurface flow, more likely surface flow.</td>
</tr>
<tr>
<td></td>
<td>Low</td>
<td>Greater than 500m from nearest water source</td>
<td>Minimum risk of contamination of water resources.</td>
</tr>
</tbody>
</table>
Figure 11.4: Preliminary risk zone map established for Gold Coast Region (Risk classifications are determined through integration of assessed criteria)
11.2 Integrated Risk Assessment

The integrated risk assessment process for the Gold Coast framework, as depicted in Figure 11.1, was conducted through the same steps as discussed in Chapter 9 and Section 10.3. This involved the assessment of the three main risk assessment processes; (1) OWTS siting and design risk; (2) Environmental Risk; and (3) Public Health Risk assessment.

11.2.1 OWTS Siting and Design Risk Assessment

Determining the OWTS siting and design risk involved characterisation of several identified hazards, including soil suitability, planning, setback distances, slope and flooding. The assessment of the identified hazards was conducted following the relevant processes explained in Chapter 9. Acceptable risk levels for these identified hazards were defined through scientific investigation as part of the research undertaken, or based on values established through current research literature identified as being suitable for providing acceptable OWTS treatment performance. Additionally, the adoption of acceptable risk levels was also discussed through consultation with the respective stakeholder groups. The acceptable risk values for each of the identified hazards in relation to the Gold Coast region are provided in Table 11.3, with the respective processes in determining the acceptable risk levels and their assessment discussed in the following sections.

Soil

Establishing the soil risks in relation to OWTS performance for the Gold Coast region required the assessment of two important relationships. Firstly, soil suitability for effluent renovation was established according to the developed soil suitability framework as discussed in Chapter 3. Additionally, it was also considered necessary to investigate the nutrient regimes (namely for nitrogen and phosphorus) and develop risk zones indicating the respective areas that retained high levels of nutrients. This could be detrimental for the use of OWTS as the excess nutrients supplied to the soil through effluent discharge could exceed the capacity of the soil to attenuate nutrients, leading contamination of groundwater and surface water.
Table 11.3: Acceptable risk levels used for assessment of OWTS hazards

<table>
<thead>
<tr>
<th>Hazard</th>
<th>Factor</th>
<th>Low</th>
<th>Medium</th>
<th>High</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil</td>
<td>Soil Suitability for effluent renovation</td>
<td>1 Kurosols and Chromosols 2 Ferrosols and Dermosols</td>
<td>3 Kandosols and Rudosols 4 Podosols and Tenosols</td>
<td>5 Organosols 6 Vertosols and Sodosols 7 Hydrosols</td>
<td>Determined according to Soil Suitability Framework (Carroll et al 2004). Utilises soil renovation ability, soil permeability and soil drainage characteristics</td>
</tr>
<tr>
<td>Nutrients</td>
<td>Nitrogen</td>
<td>&lt;0.2%</td>
<td>0.2-0.3%</td>
<td>&gt;0.3%</td>
<td>Typical soil nutrient ratings as specified in ANRA for agriculture</td>
</tr>
<tr>
<td></td>
<td>Phosphorus</td>
<td>&lt;0.003%</td>
<td>0.003-0.005%</td>
<td>&gt;0.005%</td>
<td></td>
</tr>
<tr>
<td>Planning</td>
<td>Lot Size</td>
<td>&gt; 4 Ha</td>
<td>0.4-4 Ha</td>
<td>&lt;0.4 Ha (4000m²)</td>
<td>Based on Gold Coast City Planning Scheme</td>
</tr>
<tr>
<td>Setback Distances</td>
<td>normal</td>
<td>&gt;100m</td>
<td>50-100m</td>
<td>≤ 50m</td>
<td>Specified setbacks as per On-site Sewage Code (DNRM 2002)</td>
</tr>
<tr>
<td></td>
<td>sensitive</td>
<td>&gt;200m</td>
<td>100-200m</td>
<td>≤ 100m</td>
<td>Setbacks utilised in environmentally sensitive areas</td>
</tr>
<tr>
<td>Slope</td>
<td>≤ 10%</td>
<td>&gt; 10%</td>
<td></td>
<td></td>
<td>Acceptable risk level adopted as 10%. Slope &gt; 10% at risk of causing contamination issues.</td>
</tr>
<tr>
<td>Flooding</td>
<td>Above 100 yr ARI</td>
<td>Within 1 in 100yr ARI boundary</td>
<td></td>
<td></td>
<td>Areas within the 100yr ARI at risk causing contamination, and failure of system components during flood events</td>
</tr>
</tbody>
</table>
In determining the respective soil risks, detailed investigations were conducted to evaluate the soil suitability for effluent renovation as described by Carroll et al. (2004; *Paper I*). Soil samples were collected from 73 sampling locations throughout the region, and analysed for several parameters indicated as being influential in regard to providing adequate treatment and dispersal of effluent. The soil parameters analysed and their respective influence in effluent renovation are described in *Chapters 3-5*. Figure 11.5 illustrates the spatial distribution of the soil sampling locations. These locations were selected in order to obtain a wide selection of the most common soil groups in Gold Coast region. The framework distributes the corresponding soils into several levels of suitability, as shown in Table 11.3. In order to reduce these levels to the respective low, medium and high risk zones, the resulting soil suitability indices were subsequently divided based on the following conditions; (i) soils that had medium to good renovation, with medium permeability and well drained - low risk; (ii) soils with medium renovation ability, medium to high permeability and moderately well to well drained - medium risk; and (iii) soils with poor to good renovation ability, low or high permeability, and poor to moderately well drained - high risk. The separation of the high risk soils was determined on a ‘worse case’ scenario, defined through the combination of unsuitable soil renovation ability, permeability and drainage criteria, or based on individual criteria due to extreme cases, such as very low permeability or poor drainage. The resulting risk levels associated with the defined soil types common to the Gold Coast region are provided in Table 11.3. Figure B.1 (Appendix B) highlights the developed soil risk zones for effluent renovation suitability.

The acceptable levels of nutrients within the soil were assessed on the adopted nutrient level regimes commonly used in the agricultural industry. Table 11.3 provides the acceptable nutrient levels utilised in the development of soil risk zones. To establish the risk zones for nutrient levels based on these criteria, the respective nitrogen and phosphorus concentrations obtained from the soil investigations were input into a GIS database. Nutrient risk zones were calculated by kriging (using an inverse distance weighting model) between established data points using a maximum likelihood model. These established nutrient risk zones are provided in Figure B.2 and Figure B.3 (Appendix B) respectively.
Figure 11.5: Distribution of soil samples collected throughout Gold Coast City for assessment of soil suitability for effluent renovation

The resulting cumulative soil risk zones for Gold Coast, based on the soil suitability framework and nutrient levels, were constructed by applying an overlay process (using Boolean operators) within the GIS. Essentially, for each risk zone a numeric values of 1 (low risk), 2 (medium risk) or 3 (low risk) was assigned to each developed risk layer. The corresponding values in each layer were multiplied...
together, and reclassified according to the newly established risk zones for soil as shown in Figure B.4 (Appendix. B)

**Planning**

Planning is an important factor that needs to be considered when assessing the risk associated with OWTS siting and design. Essentially, planning is related to relative lot size of the development, and consequently the respective density of OWTS. The lot size that an OWTS is to be installed has to be sufficient to allow proper treatment and dispersal of discharged wastewater. A lot size too small may not provide sufficient land to establish an effluent application area. Consequently, this may lead to effluent overloading and hydraulic failure of the application area. In addition to having adequate space for adequate use of OWTS, the size of the respective lot must also allow sufficient room to incorporate an appropriate setback distance. This is to prevent any pollutant contamination from leaving the lot itself. This issue is related to all of the site and soil characteristics that need to be assessed prior to installation of an OWTS. Lot size is also related to the density of OWTS in the surrounding area. Smaller lot sizes will enable higher housing densities, and consequently increase the number of OWTS in unsewered developments.

The corresponding risk levels associated with planning and lot size were established according to the current Gold Coast City Council Planning Scheme. Lot sizes of area less than 4000m² (0.4Ha) were deemed to induce a significant risk related to poor treatment performance issues and high densities of OWTS. Areas between 0.4 to 4Ha were considered to present a medium risk, with areas greater that 4Ha considered low risk. These corresponding risk levels were implemented into the GIS, and a map showing the risk zones for planning was developed as shown in Figure B.5, (Appendix B).

**Slope**

The significance of slope in relation to poor OWTS performance is associated in three main ways. Steep slopes increase the amount of runoff produced, which in turn increases the potential risk of contamination of surface water. Hydraulic failure or surfacing of effluent from subsurface systems, as well as the use of surface irrigation, can be a concern in areas of high slope for this reason. Secondly, the use of
subsurface application areas adjacent to steep slopes may cause lateral flow of effluent to take place through the soil layers resulting in seepage of effluent down slope of the dispersal area.

Typically, slopes in the range of 6-10% are generally viewed as suitable for surface irrigation systems, with steeper slopes contributing to higher runoff volumes and hence higher risks of pollution (Kleene et al 1993, AS1547:2000, Wells 2001). For subsurface dispersal systems, slopes of 10-20% are commonly accepted as the norm (Brouwer 1983, AS1547:2000, US EPA 2002), with slopes greater than 20% considered inappropriate for providing adequate dispersal of effluent. This can result in failure of the subsurface system leading to surfacing of effluent.

Consequently, the threshold value for acceptable risk was taken as 10%, being the maximum slope that will have minimal effect from surface irrigation systems, and the minimum slope for subsurface dispersal systems to provide adequate renovation of effluent. Slopes less than 10% were deemed as low risk, with steeper slopes classified as high risk, leading to inadequate effluent renovation and contamination issues. To develop the respective risk zones for slope, several GIS operations had to be undertaken. Firstly, a digital elevation model (DEM) was constructed for the Gold Coast region based on 10 metre contours. The developed DEM was then reclassified based on the adopted acceptable risk profiles to spatially identify the areas with slopes greater than 10%. The resulting risk map is shown in Figure B.6 (Appendix B).

**Setback Distances**
Ensuring appropriate separation or setback distances between the on-site system and nearby water sources, is a crucial issue. Setback distances are included in current standards and guidelines to minimise potential environmental and public health risks due to poor OWTS performance. Setback distances implicitly include risk-based management ideals into current performance standards. The current setback distances stipulated in the *On-site Sewage Code* (DNRM 2002) recommends that a horizontal distance of 50m (primary effluent) between the system and adjacent water sources be used, and a vertical distance of 1.2m (primary effluent) between the dispersal field and the water table. Reduced setbacks have been specified for secondary treated
Chapter 11.0 Implementation of Integrated Risk Framework, Gold Coast City

effluent. However, these values typically evolve around public health issues with distances based on viral transport and fate models. Although indicating that these distances may be sufficient for protection of public health, in some cases, environmental risks may be significantly higher than the public health issues. An initial assessment of the potential for increasing setback distances was undertaken through this research, and the outcomes were discussed through a published conference paper (Appendix A). Through this study, it was found that the ability of nutrients, in particular nitrate, to move from high densities of OWTS would be significantly higher than that of the adopted setbacks obtained through viral die off models.

The assessment of risk associated with setback distances was established based on the currently adopted criteria stipulated in AS1547:2000 and the Queensland On-site Sewerage Code (DNRM 2000). In the determination of risk zones, areas within a distance of 50m were considered to have a high risk and those within a distance between 50 – 100m was considered to be of medium risk. However, for areas found to have a high environmental risk, the current setback distances were considered inadequate for protection of environmental values. Hence, it was considered necessary to increase the setbacks distances in these areas to demarcate high and medium risk zones. For these particular cases, setback distances were increased to 100m horizontal distance for high risk, and 100-200m for medium risk areas. These setback distances were used to develop buffer zones surrounding the major water resources within the Gold Coast region, as shown in Figure B.7 (Appendix B).

**Flooding**

Flooding of treatment system and dispersal areas can be a significant issue depending on several factors. Firstly, if an effluent dispersal area is flooded, there is a strong possibility of contamination of the flood waters. However, this is dependent on the average recurrence interval (ARI) of the particular flood event. Large floods will essentially dilute the exposed effluent, minimising the potential risks associated with contamination. This may not be the case with smaller flood events, with flooding of dispersal areas allowing the transportation of effluent pollutants off-site. However, large events will flood larger areas, and therefore higher numbers of OWTS may be affected. This may increase the potential risk by reducing any dilution effects as a
result of the higher numbers of OWTS exposed to the flood event. The most commonly adopted flood level for assessing the potential risk from OWTS is the 100 year ARI. Adopting this flood level as the acceptable risk also inherently includes lower flood levels within the identified risk areas. Very few studies have investigated the probability of contamination from OWTS as a result of flood events. It can be difficult, if not impossible, to collect representative data during flood events.

Therefore, in assessing the risks associated with flood events, the 100yr ARI flood boundary was used as the acceptable risk level. The region within this boundary was considered at high risk, with outside regions at low risk. The respective risk zones developed for flood effects for the Gold Coast region are provided in Figure B.8 (Appendix B).

11.2.2 Environmental and Public Health Risk

The assessment of both, environmental and public health risk was established based on assessing the risk of contamination at monitored locations. This was developed around an engineering risk analysis approach as outlined by Ganoulis (1994). The risk established through this process is determined by the probability of failure of contaminant concentrations failing acceptable threshold concentrations and is equivalent to:

\[
Risk = probability\ of\ failure = P_f = P(L > R) = \int_0^\infty \left( \int_0^\infty f_{LR}(L,R) dR \right) dL
\]

(2)

where \( L \) = pollutant loading or concentration and \( R \) = resistance or prescribed water quality standard or threshold. More detailed discussion of this risk assessment process is provided in Chapter 9. The specified water quality parameters for environmental (focusing on nitrate and phosphate) and public health (faecal coliforms and \( E. coli \)) risk assessment were obtained from monitored groundwater and surface water locations throughout Gold Coast region. Water samples were collected on a fortnightly basis over a four month period and analysed for the respective contaminants. The monitoring locations used for sampling are shown in Figure 11.6. The risk (probability) of monitored contaminants failing to meet the specified water quality guidelines for both drinking water (NHMRC 1996) and
recreational water and aesthetics (recreational, primary and secondary contact) (ANZECC 2000) was determined. Respective probabilities were established according to fitted probability distribution functions based on the analysed data for each assessed area. For an area to have a low environmental or public health risk, the arithmetic mean of the assessed contaminants had to be lower than the stipulated guideline level 100% of the time. The area is considered at risk whenever the guideline is exceeded.

Environmental risk was determined based on the assessment of nitrate and phosphate contamination of surface and groundwater. Investigation of groundwater and surface water quality at Jacobs Well and Cabbage Tree Point has been discussed in Chapter 6. However, several other areas as depicted in Figure 11.6 were also monitored for the assessment of environmental and health risks. Analysed water quality data for all monitored locations are provided in Appendix C. The adopted threshold values used in the risk assessment were those specified in the ANZECC (2000) water quality guidelines for aesthetics. This also relates to the limiting concentrations needed to cause nutrient enrichment and subsequent eutrophication of water resources and algae blooms. Table 11.4 provides the adopted guideline values employed in the environmental risk assessments. In establishing the respective probabilities for failing the threshold values, cumulative lognormal probability functions for nitrate and phosphate were fitted to the analysed data, as depicted in Figure 11.7a – d. Table 11.5 provides the determined probabilities for the corresponding monitoring locations.

Environmental risk maps were developed based on the quantified risks for the monitored locations. Those areas that failed to meet the specified water quality guideline threshold values were considered an environmental risk. The respective probabilities for these monitored areas were projected to other unmonitored areas that retained similar soil, site and planning characteristics in order to establish risk profiles for the entire Gold Coast region. The developed environmental risk maps are provided in Figure B.9 (Appendix B).
Figure 11.6: Groundwater and surface water monitoring locations used for assessing environmental and public health risks
### Table 11.4: Concentrations threshold values used for risk assessment

<table>
<thead>
<tr>
<th>Issue</th>
<th>Parameter</th>
<th>Response</th>
<th>Guideline values (threshold)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Environmental</td>
<td>NO₃⁻ N</td>
<td>General Water Quality</td>
<td>10mg/L</td>
<td>ANZECC (2000)</td>
</tr>
<tr>
<td></td>
<td>PO₄³⁻ P</td>
<td>General Water Quality</td>
<td>No Guidelines</td>
<td>ANZECC (2000)</td>
</tr>
<tr>
<td></td>
<td>Eutrophication*</td>
<td>≤ 50µg/L – Freshwater Rivers ≤ 30µg/L – Estuaries</td>
<td>ANZECC (2000)</td>
<td></td>
</tr>
<tr>
<td>Faecal Coliforms</td>
<td>Drinking water</td>
<td>0 cfu/100mL</td>
<td>NHMRC (1996) ANZECC (2000)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Primary Contact (recreation, swimming)</td>
<td>≤ 150 cfu/100mL</td>
<td>ANZECC (2000)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Secondary Contact (irrigation, boating)</td>
<td>≤ 1000 cfu/100mL</td>
<td>ANZECC (2000)</td>
<td></td>
</tr>
<tr>
<td>Public Health</td>
<td>E.coli</td>
<td>Drinking water</td>
<td>0 cfu/100mL</td>
<td>NHMRC (1996) ANZECC (2000)</td>
</tr>
<tr>
<td></td>
<td>Primary Contact (recreation, swimming)</td>
<td>≤ 150 cfu/100mL</td>
<td>ANZECC (2000)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Secondary Contact (irrigation, boating)</td>
<td>≤ 1000 cfu/100mL</td>
<td>NHMRC (1996) ANZECC (2000)</td>
<td></td>
</tr>
<tr>
<td>NO₃⁻ N</td>
<td>Drinking (ingestion)</td>
<td>10mg/L</td>
<td>NHMRC (1996) ANZECC (2000)</td>
<td></td>
</tr>
</tbody>
</table>
Figure 11.7a: Cumulative probability distributions for nitrate at groundwater monitoring locations

Figure 11.7b: Cumulative probability distributions for phosphate for groundwater monitoring locations
Figure 11.7c: Cumulative probability distributions for nitrate for surface water monitoring locations

Figure 11.7d: Cumulative probability distributions for phosphate for surface water monitoring locations
Table 11.5: Calculated probabilities of nutrients exceeding adopted water quality guidelines

<table>
<thead>
<tr>
<th>Location</th>
<th>Source</th>
<th>NO₃⁻ (10 mg/L)</th>
<th>CDF²</th>
<th>P(c&gt;t) %</th>
<th>PO₄³⁻ (50 μg/L (Fresh), 30 μg/L (Marine))</th>
<th>CDF</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Mean, μ mg/L</td>
<td>σ</td>
<td>CDF</td>
<td>Mean, μ mg/L</td>
<td>σ</td>
</tr>
<tr>
<td>Cabbage Tree</td>
<td>Ground</td>
<td>23.140</td>
<td>19.978</td>
<td>77.4</td>
<td>4.195</td>
<td>5.387</td>
</tr>
<tr>
<td></td>
<td>Surface</td>
<td>6.248</td>
<td>3.011</td>
<td>L</td>
<td>2.665</td>
<td>4.437</td>
</tr>
<tr>
<td>Jacobs Well</td>
<td>Ground</td>
<td>43.734</td>
<td>46.214</td>
<td>89.8</td>
<td>16.075</td>
<td>7.408</td>
</tr>
<tr>
<td></td>
<td>Surface</td>
<td>8.623</td>
<td>4.311</td>
<td>N</td>
<td>5.675</td>
<td>5.177</td>
</tr>
<tr>
<td>Coomera</td>
<td>Ground</td>
<td>58.953</td>
<td>57.110</td>
<td>96.2</td>
<td>17.985</td>
<td>8.967</td>
</tr>
<tr>
<td></td>
<td>Surface</td>
<td>9.260</td>
<td>4.944</td>
<td>L</td>
<td>4.246</td>
<td>1.944</td>
</tr>
<tr>
<td>Bonogin</td>
<td>Ground</td>
<td>58.530</td>
<td>49.890</td>
<td>97.8</td>
<td>8.190</td>
<td>8.551</td>
</tr>
<tr>
<td></td>
<td>Surface</td>
<td>7.323</td>
<td>2.711</td>
<td>N</td>
<td>11.192</td>
<td>5.899</td>
</tr>
<tr>
<td>Beechmont</td>
<td>Ground</td>
<td>8.745</td>
<td>3.080</td>
<td>34.2</td>
<td>3.257</td>
<td>2.216</td>
</tr>
<tr>
<td>Tallebudgera</td>
<td>Surface</td>
<td>6.929</td>
<td>2.484</td>
<td>10.8</td>
<td>14.951</td>
<td>7.674</td>
</tr>
</tbody>
</table>

1: ARL: Action Limit
2: CDF: Cumulative Distribution Function

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The assessment of public health risk from OWTS involved two major stages. Firstly, collected groundwater and surface water samples from monitored locations (Figure 11.6) were analysed for faecal coliforms (FC) and confirmed *Escherichia coli* (*E. coli*). An initial public health risk assessment was conducted on the obtained *E. coli* counts against the acceptable threshold values for drinking water and for primary and secondary contact, as outlined in NHMRC (1996) and ANZECC (2000). This allowed an assessment based on faecal indicators which are typically used for assessing faecal contamination of water resources. Cumulative log normal probability distribution functions were fitted to the collected *E. coli* data from monitored locations and are provided in Figure 11.8.

![CDF E. coli Groundwater](image)

**Figure 11.8a**: Cumulative probability distributions for *E. coli* for groundwater monitoring locations
Figure 11.8b: Cumulative probability distributions for E. coli for surface water monitoring locations

The calculated risks (probabilities) for public health based on E. coli counts are given in Table 11.6. As indicated, all monitored locations, for both surface and groundwater, failed to meet the specified guideline for drinking water. However, this is not a major concern as the water resources in the monitored locations are used primarily for recreational and gardening purposes. Therefore, the primary and secondary contact guidelines are more appropriate for assessing the public health risk. However, as indicated in Chapters 7 and 8, the actual source of faecal contamination based on simple faecal indicators such as FC and E. coli, may have originated from several sources. This would inevitably increase the level of uncertainty in the risk assessment, and lead to an over estimation of the derived risk. Consequently, the use of ARP to source track human faecal contamination, and subsequently link this to OWTS was conducted.

The use of ARP for two monitored catchments at Bonogin and Tallebudgera Creeks were assessed and detailed descriptions of the scientific process used, and the development and predicative capability of the ARP source database are described in Chapter 8. However, several other monitored locations were also assessed using ARP, and the identified sources of faecal contamination at each of the monitoring locations
are provided in Appendix D. The use of ARP allows a reduced uncertainty in assessing public health risks compared to using general faecal indicators, due to the identification of the respective sources. The percentage of human *E. coli* was subsequently used to assess the risk to public health from OWTS. This was achieved by reducing the number of *E. coli* in the analysed samples by the identified percentage of human source *E. coli*. The corresponding public health risks established on this basis are given in Table 11.6. Public Health risk maps were developed in the same manner to that for environmental risks, with the resulting risk areas indicated in Figure B.10 (Appendix B).

**11.3 Development of Integrated Risk Maps**

Development of the final integrated risk map for the Gold Coast region involved several procedures developed through the use of GIS technology. The GIS employed for this research project included MAPINFO version 7.0, a vector based GIS system, and ARCVIEW version 8.0, a raster based GIS. MAPINFO is the commonly adopted system that most local authorities, including GCCC, utilise for keeping databases related to planning, design and assessment. Hence, most of the available information obtained from GCCC for this research was in MAPINFO format. However, the ability of MAPINFO for allowing overlays of spatial information and arithmetic interpretation of spatial data is limited and most of the processes required for developing the integrated risk maps would require substantial manual intervention if MAPINFO was to be used for integrating the risk layers. Therefore, ARCVIEW software was used for the initial construction of the risk maps, due to its superior analytical tools and algorithms for overlaying and reclassifying. The final risk map was subsequently converted to MAPINFO format.

The procedure for developing the integrated risk map involved overlaying each individual layer obtained through the assessment of the various risks, established during the risk assessment stage. The individual GIS layers included (1) soil; (2) planning (Lot size); (3) setback distances; (4) slope; (5) flooding; (6) public health risk; and (7) environmental risk. Layers (1) to (5) were first combined using boolean
Table 11.6 Calculated probabilities of Total and human *E. coli* exceeding adopted water quality guidelines

<table>
<thead>
<tr>
<th>Location</th>
<th>Source</th>
<th>Mean, μ mg/L</th>
<th>Mean, μ mg/L</th>
<th>St. Dev. σ</th>
<th>CDF4 (c&gt;t) %</th>
<th>P(c&gt;t) %</th>
<th>P(c&gt;t) %</th>
<th>P(c&gt;t) %</th>
<th>P(c&gt;t) %</th>
<th>P(c&gt;t) %</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cabbage</td>
<td>Tree</td>
<td>152</td>
<td>586</td>
<td>L</td>
<td>100.0</td>
<td>31.0</td>
<td>33.2</td>
<td>10.3</td>
<td>1.0</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>Point</td>
<td>170</td>
<td>250</td>
<td>L</td>
<td>100.0</td>
<td>63.4</td>
<td>33.7</td>
<td>21.4</td>
<td>6.4</td>
<td>1.4</td>
</tr>
<tr>
<td>Jacobs Well</td>
<td>Ground</td>
<td>72</td>
<td>105</td>
<td>L</td>
<td>100.0</td>
<td>35.1</td>
<td>5.9</td>
<td>2.1</td>
<td>1.0</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>Surface</td>
<td>45</td>
<td>79</td>
<td>L</td>
<td>100.0</td>
<td>16.7</td>
<td>5.4</td>
<td>0.9</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Coomera</td>
<td>Ground</td>
<td>564</td>
<td>700</td>
<td>L</td>
<td>100.0</td>
<td>77.6</td>
<td>81.3</td>
<td>63.1</td>
<td>14.4</td>
<td>9.1</td>
</tr>
<tr>
<td></td>
<td>Surface</td>
<td>329</td>
<td>567</td>
<td>L</td>
<td>100.0</td>
<td>66.7</td>
<td>53.3</td>
<td>35.6</td>
<td>6.3</td>
<td>2.2</td>
</tr>
<tr>
<td>Bonogin</td>
<td>Ground</td>
<td>250</td>
<td>631</td>
<td>L</td>
<td>100.0</td>
<td>38.2</td>
<td>36.5</td>
<td>13.9</td>
<td>4.6</td>
<td>0.6</td>
</tr>
<tr>
<td></td>
<td>Surface</td>
<td>164</td>
<td>147</td>
<td>L</td>
<td>100.0</td>
<td>42.4</td>
<td>39.4</td>
<td>16.7</td>
<td>5.3</td>
<td>0.9</td>
</tr>
<tr>
<td>Beechmont</td>
<td>Ground</td>
<td>1</td>
<td>1</td>
<td>L</td>
<td>100.0</td>
<td>1.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Tallebudgera</td>
<td>Surface</td>
<td>101</td>
<td>60</td>
<td>L</td>
<td>100.0</td>
<td>35.8</td>
<td>16.0</td>
<td>5.7</td>
<td>5.0</td>
<td>0.3</td>
</tr>
</tbody>
</table>
overlays to produce the risk map for OWTS siting and design risks. This was then overlaid with the public health and environmental risk maps to produce the final risk map. A visualisation of the process utilised in obtaining the respective maps, and the maps developed for each of the risk processes is provided in Figure B.11a and B.11b (Appendix B). Each level of risk for each hazard was assigned a numeral value of 1 (low risk), 2 (medium risk) or 3 (low risk). Using an arithmetic overlay procedure, the corresponding values in each layer were multiplied together and reclassified back to the initial numerical values for identifying high, medium and low risk. For implementing the developed integrated risk map for the Gold Coast region, identified risk areas were additionally reduced to two major classifications, ‘low risk’ and ‘at risk’ (medium and high risk) areas. This was undertaken to remove any discrepancies between the determination of medium and high risks. Effectively, this follows the same probability of failure, whereby an area at risk of causing environmental or public health impacts is required to undergo an increased level of assessment. The final combined integrated risk map for the Gold Coast region is presented in Figure 11.9. This map, along with the risk assessment criteria was implemented as part of the GCCC planning scheme.

11.4 Assessment of at risk areas

Once the integrated risk zones for OWTS were established, the identification of suitable assessment techniques for the appropriate assessment of at risk areas was developed. Where the area of assessment falls within a low risk area, the currently adopted standards and codes were considered as appropriate for the assessment and management OWTS. This generally implies AS/NZS 1547:2000, the Australian Standard for On-site Sewage Systems, as well as the locally adopted codes and guidelines. However, as shown in the developed framework (Figure 11.1) if a low risk area is rezoned for future development, this may subsequently increase the risk level as a result of reduced lot sizes, and additional assessment based on the planning requirements should be undertaken. For ‘at risk’ areas, the additional assessment requirements are determined according to the identified limiting hazards specified through the assessed risk areas. Table 11.7 provides the limiting hazards that are associated with the respective ‘at risk’ regions, and the necessary requirements for
additional assessment. These limiting criteria are highlighted in the developed GIS data base and resulting risk map, to indicate what additional assessment is required within the categorised risk areas.

Figure 11.9: Final combined integrated risk map for the Gold Coast region
Table 11.7: Limiting Hazards for at risk areas, Gold Coast City

<table>
<thead>
<tr>
<th>Hazard</th>
<th>Factor</th>
<th>Notation</th>
<th>Risk Limitations</th>
<th>Limiting Hazard</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil (S)</td>
<td>Renovation ability</td>
<td>r</td>
<td>1 and 2¹</td>
<td>3 to 7¹</td>
</tr>
<tr>
<td></td>
<td>Permeability</td>
<td>p</td>
<td>0.005-0.5 m/d</td>
<td>&lt;0.005 or &gt;0.5 m/day</td>
</tr>
<tr>
<td></td>
<td>Drainage</td>
<td>d</td>
<td>Moderate to well drained</td>
<td>Imperfectly to poorly drained, permanent or seasonal saturation</td>
</tr>
<tr>
<td></td>
<td>Nutrients</td>
<td>n</td>
<td>TN ≤ 0.3%</td>
<td>TN &gt; 0.3%</td>
</tr>
<tr>
<td></td>
<td>Planning (P)</td>
<td>l</td>
<td>&gt; 4.0 Ha</td>
<td>&lt; 4.0 Ha</td>
</tr>
<tr>
<td></td>
<td>Setback (B)</td>
<td></td>
<td>&gt; 50m</td>
<td>&lt; 50m</td>
</tr>
<tr>
<td></td>
<td>Slope (G)</td>
<td></td>
<td>&lt; 10%</td>
<td>&gt; 10%</td>
</tr>
<tr>
<td></td>
<td>Flood (F)</td>
<td></td>
<td>Above 1 in 100 yr ARI²</td>
<td>Below 1 in 100 yr ARI²</td>
</tr>
<tr>
<td>Public Health (H)</td>
<td>Mean Human³ E.coli contaminants below threshold⁴</td>
<td>Probability mean Human³ E.coli contaminants exceed threshold⁴</td>
<td>Contamination of water resources with pathogenic organisms, risk to public health</td>
<td></td>
</tr>
<tr>
<td>Environment (E)</td>
<td>Mean nutrient contaminants below threshold⁴</td>
<td>Probability mean nutrient contaminants exceed threshold⁴</td>
<td>Contamination of water resources with pathogenic organisms, risk to environment, nutrient enrichment, algae blooms</td>
<td></td>
</tr>
</tbody>
</table>

¹ Determined according to Soil suitability for effluent renovation framework (Carroll et al 2004)
² Average Recurrence Interval
³ Determined by source tracking using Antibiotic Resistance Pattern analysis
⁴ Stipulated thresholds for drinking water, recreational and aesthetics water quality (NHMRC 1996 and ANZECC 2000)
11.5 Critical Point Monitoring for Risk Management

The management of the inherent risks characterised through the integrated risk framework for the Gold Coast region involved two main processes. Firstly, management of OWTS was achieved by developing new and more appropriate assessment guidelines as outlined in Section 10.2.4. This was effectively aimed at mitigating the possible risk inherent in the use of OWTS in areas that are not adequate in meeting the specified requirements set out in the standards and guidelines. Secondly, a critical point monitoring (CPM) program to allow GCCC to monitor the ‘at risk’ areas for identified critical parameters was established. In a risk management context, CPM identifies the critical points within a management system that should be monitored to provide suitable mitigation of identified risks. Figure 11.10 provides the steps generally employed in a CPM program. In constructing the integrated risk framework, the critical points associated with the ‘at risk’ areas were already identified. Therefore, the incorporation of the CPM program was easily integrated into the overall framework. The critical parameters used for the CPM program were identified through the research undertaken and included parameters focusing on soil and site requirements and environmental and public health issues. Table 11.8 highlights the critical parameters established for sensitive areas identified through the respective data analysis. Essentially, these are based on the limiting factors isolated through the risk assessment stage for the different ‘at risk’ areas.

Integration of the developed integrated risk framework and risk maps into the Gold Coast City planning scheme guidelines will enable the Gold Coast City Council to determine which regions within their jurisdictional area are at risk of causing detrimental environmental and public health impacts. This will enable GCCC to incorporate best management practices in a more practical and efficient manner specifically targeting ‘at risk’ regions, and thereby mitigating the inherent risks.
Table 11.8: Critical parameters to be monitored as part of CPM for Gold Coast City

<table>
<thead>
<tr>
<th>Location</th>
<th>Resource</th>
<th>Critical Parameter to be Monitored</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cabbage Tree Point</td>
<td>Groundwater</td>
<td>Nitrate (NO\textsubscript{3}^-), \textit{E.coli}</td>
</tr>
<tr>
<td></td>
<td>Surface Water</td>
<td>\textit{E.coli}</td>
</tr>
<tr>
<td>Jacobs Well</td>
<td>Groundwater</td>
<td>Nitrate (NO\textsubscript{3}^-), Phosphate (PO\textsubscript{4}^{3-})</td>
</tr>
<tr>
<td></td>
<td>Surface Water</td>
<td>Nitrate (NO\textsubscript{3}^-)</td>
</tr>
<tr>
<td>Coomera</td>
<td>Groundwater</td>
<td>Nitrate (NO\textsubscript{3}^-), Phosphate (PO\textsubscript{4}^{3-}) and \textit{E.coli}</td>
</tr>
<tr>
<td></td>
<td>Surface Water</td>
<td>Nitrate (NO\textsubscript{3}^-), Phosphate (PO\textsubscript{4}^{3-}) and \textit{E.coli}</td>
</tr>
<tr>
<td>Bonogin</td>
<td>Groundwater</td>
<td>Nitrate (NO\textsubscript{3}^-), \textit{E.coli}</td>
</tr>
<tr>
<td></td>
<td>Surface Water</td>
<td>Phosphate (PO\textsubscript{4}^{3-}), \textit{E.coli}</td>
</tr>
<tr>
<td>Tallebudgera</td>
<td>Surface Water</td>
<td>Phosphate (PO\textsubscript{4}^{3-}), \textit{E.coli}</td>
</tr>
<tr>
<td>Lower Beechmont</td>
<td>Groundwater</td>
<td>Nitrate (NO\textsubscript{3}^-)</td>
</tr>
</tbody>
</table>
CHAPTER 12.0

CONCLUSIONS AND RECOMMENDATIONS

12.1 Conclusions

Due to the inherent treatment performance issues associated with OWTS, failure is a common scenario and can lead to environmental and public health concerns. Typically, these issues are related to the current assessment of site and soil characteristics, and the general management of OWTS. In order to obtain sustainable wastewater treatment through OWTS, more scientifically robust methods of assessing site suitability is needed. The main focus of this research project was to develop an integrated risk-based approach for assessing and managing on-site wastewater treatment systems. The research aims and objectives were achieved through the development of three primary fundamental processes. These were;

- Hazard identification and characterisation
  
  Hazard identification and characterisation was conducted through several workshops held in conjunction with Gold Coast City Council and the invited stakeholders. This allowed the key hazards of most concern associated with OWTS performance to be identified. These key hazards were subsequently characterised as part of the integrated risk assessment process.

- Development of an integrated risk framework
  
  The development of the individual risk assessments and integrated risk framework required resolving several inherent deficiencies evident in the current standards and codes for the assessment and management of OWTS. These were achieved through the scientific investigations detailed in Chapters 3-9. The investigations conducted focused on three key research areas; (i) assessment of soil suitability for effluent renovation; (ii) assessment of contamination of ground and surface waters as a result of OWTS failure; and (iii) integrated assessment of OWTS siting and design, environmental and public health risks. These studies
involved extensive field investigations, sampling and laboratory testing and detailed data analysis.

- Development of a Critical Point Monitoring program for risk management
  To establish a risk management process to provide an appropriate management regime for OWTS within the study region, a critical point monitoring program (CPM) was formulated. This entailed the identification of the key critical parameters that contribute to the characterised risks at monitored locations within the study area. This was achieved through the research investigations described in Chapters 6 and 7. The CPM allows more direct managerial procedures to be implemented, targeting the specific hazards at sensitive areas throughout Gold Coast Region.

In achieving the specified research aims and objectives, several detailed investigations were conducted to resolve the identified deficiencies lacking in the current assessment and management techniques. The assessment of soil suitability for providing adequate treatment and dispersal of effluent is one of the most important aspects related to the proper performance of OWTS. The ability of the soil for providing adequate attenuation and removal of effluent contaminants is dependent on several factors which need to be adequately assessed to ensure the soil is capable of providing adequate treatment of effluent. Therefore, in order to ensure that proper treatment and dispersal of effluent from OWTS occurs, it is imperative that assessment of soil suitability for effluent renovation be conducted in a scientific manner. To achieve this, a comprehensive understanding of the site and soil characteristics that influence the effluent treatment process was obtained, with the scientific principles and processes used in developing the criteria for assessing site and soil suitability. This was the main focus of the scientific papers in Chapters 3-5. The developed soil suitability framework provides a process for determining the suitability of soil for providing appropriate renovation of effluent. Although the soils investigated in developing the framework were restricted to Southeast Queensland, the processes and techniques used for assessing soil suitability have been developed on a generic basis.
Chapter 12.0 Conclusions and Recommendations

The contamination of water resources as a result of poorly performing OWTS is a widely recognised issue. However, research relating to the various factors that can influence the fate and transport of contaminants through the environment have generally been neglected. The movement of pollutants from OWTS into the groundwater, and eventually surface water are reliant on a number of physical and chemical processes which may either hinder or advance contaminant transport. Investigations into the different factors and their influence on the transport of contaminants were undertaken through this research (Chapter 6 and 7). It was found that high densities of OWTS and their proximity to water resources greatly influenced contaminant concentrations in ground and surface waters. However, factors such as soil type and slope were also related to the respective level of contaminants, although these were found to be more influential on contaminant fate. Statistical analysis of collected water samples also indicated that the cumulative effect of multiple investigated factors was more influential on the fate and transport of pollutants within the environment than any singular factor. This is particularly important for the management of environmental and public health risks, as these factors need to be considered when implementing management strategies to mitigate the respective risks.

Additionally, an investigation into the various techniques available for sourcing nircoorganisms back to OWTS was undertaken. Through this research, antibiotic resistance pattern (ARP) analysis was utilised to source track enumerated E. coli isolates obtained from monitored water sources, to their host of origin (Chapter 8). This research proved particularly useful in determining human from non-human faecal contamination and subsequently allowed the identification of areas which were highly sensitive to OWTS and indicated significant public health risk.

In order to overcome the inherent risks associated with the poor performance of OWTS, a more universal and scientifically robust assessment and management framework is necessary. The developed integrated risk framework (Chapter 9) is specifically aimed at assessing and managing the inherent environmental and public health risks associated with OWTS. As detailed through the case study for Gold Coast region, detailed investigations have allowed the incorporation of scientific information into the assessment of OWTS siting and design and environmental and
public health risks. This information, along with the resulting risks, was developed into a GIS database to allow spatial identification of at risk areas. This allowed more appropriate management protocols to be implemented aimed specifically at mitigating the inherent environmental and public health risks associated with poor OWTS performance within the identified at risk areas.

### 12.2 Recommendations

Although the developed integrated risk framework has helped to strengthen the current standards and codes, there still remain two critical areas that have not been addressed through this research. Therefore, further research towards these areas is recommended to increase the effectiveness of the developed risk framework. These include:

- Assessment and implementation of appropriate setback distances based on site specific investigations rather than applying generalised setbacks for all scenarios. As described throughout the current research, assessment of site suitability is dependent on a number of soil and site related characteristics which can substantially differ from one location to the next. As such, setback distances need to be investigated based on these specific site characteristics rather than adopting standardised distances for all site conditions.

  Additionally, current setback distances focus specifically on public health issues. Although this is a critical factor, some areas may be more prone to environmental impacts. The resulting adverse impacts due to the transport of nutrients can be far in excess to that of public health impacts. Therefore, an investigation into the development of setback distances based on environmental issues also needs to be addressed.

- Density is an important issue relating to the management of OWTS. Increased densities of OWTS are well recognised for increasing the potential for contamination of water resources, particularly groundwater. Additionally, from the research undertaken, it was found that high OWTS densities on suitable soil
did not impact on water resources to the extent as that for unsuitable soil. As such, the acceptable density of OWTS will differ depending on the respective soil and site characteristics. However, there currently exists no scientifically defensible information that details what density of OWTS can be adopted based on soil conditions, without causing detrimental impacts on water resources.
CHAPTER 13.0

CONSOLIDATED LIST OF REFERENCES


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Chapter 13.0 References


growth centres in developing countries. *Journal of Cleaner Production* **10**: 271-281.


Chapter 13.0 References


APPENDICIES
APPENDIX A

ASSESSMENT OF ENVIRONMENTAL AND PUBLIC HEALTH RISK OF ON-SITE WASTEWATER TREATMENT SYSTEMS

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On-site Wastewater Treatment X: Proceedings of the Tenth National Symposium on Individual and Small Community Sewage Systems. ASAE, Sacramento, California, USA. pp. 368-376

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APPENDIX B

RISK ZONES DEVELOPED FOR GOLD COAST CITY
Figure B.1: Soil suitability for effluent renovation risk classifications established for Gold Coast City.

Figure B.2: Nitrogen risk classifications established for Gold Coast City.
Appendix B
Risk Zones Developed for Gold Coast City

Figure B.3: Phosphorus risk classifications established for Gold Coast City

Figure B.4: Cumulative soil and nutrient risk classifications for Gold Coast City
Figure B.5: Planning scheme (lot size) risk classifications for Gold Coast City

Figure B.6: Slope risk classifications for Gold Coast City
Figure B.7: Setback distance risk classifications for Gold Coast City

Figure B.8: Flooding risk classifications for Gold Coast City
Appendix B
Risk Zones Developed for Gold Coast City

Figure B.9: Environmental risk classifications for Gold Coast City

Figure B.10: Public health risk classifications for Gold Coast City
Figure B.11a Public health risk classifications for Gold Coast City
Figure B.11b Public health risk classifications for Gold Coast City
Appendix C

Water Quality Data from Monitored Sampling Locations, Gold Coast City
### Appendix C

**Water Quality Data from Monitored Sampling Locations**

Average (range) concentrations of water quality data from sample monitoring locations

Table C.1: Water quality parameters analysed and notation

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### Water Quality Data from Monitored Sampling Locations

#### Table C.2 (continued) Average (range) concentrations of chemical contaminants at sample monitoring locations

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### Appendix C

#### Water Quality Data from Monitored Sampling Locations

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<td>(100-8800)</td>
<td>(100-300)</td>
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<td>13383</td>
<td>2017</td>
</tr>
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<td></td>
<td>(2400-660000)</td>
<td>(1000-48000)</td>
<td>(100-7000)</td>
</tr>
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<td>1052</td>
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<td>(45-1700)</td>
<td>(1-100)</td>
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</tr>
<tr>
<td></td>
<td>(2-200)</td>
<td>(5-200)</td>
<td>(2-2)</td>
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<tr>
<td>TA1</td>
<td>1244</td>
<td>1044</td>
<td>108</td>
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<td></td>
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<td>(180-1900)</td>
<td>(40-160)</td>
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<tr>
<td>TA2</td>
<td>1338</td>
<td>1900</td>
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</tr>
<tr>
<td></td>
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<td>(1100-3000)</td>
<td>(50-180)</td>
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<td>TA3</td>
<td>2544</td>
<td>2080</td>
<td>86</td>
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<tr>
<td></td>
<td>(18-6000)</td>
<td>(1200-3800)</td>
<td>(40-170)</td>
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APPENDIX D

SOURCING OF FAECAL CONTAMINATION

GOLD COAST CITY
Identification of *E. coli* isolates using Antibiotic Resistance Pattern Analysis

For methodology in using ARP for source identification, see Chapter 8.0

Table D.1 Antibiotics and concentrations used for determining ARP patterns

<table>
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<tr>
<th>Antibiotic</th>
<th>Concentrations (mg/mL)</th>
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<tbody>
<tr>
<td>Amoxicillin</td>
<td>5 10 15 20</td>
</tr>
<tr>
<td>Cephalothin</td>
<td>10 25 50 100</td>
</tr>
<tr>
<td>Erythromycin</td>
<td>20 50 100 200</td>
</tr>
<tr>
<td>Gentamicin</td>
<td>20 40 60 80</td>
</tr>
<tr>
<td>Ofloxacin</td>
<td>5 10 15 20</td>
</tr>
<tr>
<td>Chlortetracycline</td>
<td>20 40 60 80</td>
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<td>Tetracycline</td>
<td>20 40 60 80</td>
</tr>
<tr>
<td>Moxalactam</td>
<td>5 10 15 20</td>
</tr>
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</table>

Table D.2: Known source isolate groups and assigned group categories

<table>
<thead>
<tr>
<th>Source</th>
<th>No. Isolates</th>
<th>ARP 1 Category</th>
<th>ARP 2 Category</th>
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<tbody>
<tr>
<td>Human</td>
<td>210</td>
<td>Human</td>
<td>Human (HUM)</td>
</tr>
<tr>
<td>Dog</td>
<td>178</td>
<td>Human</td>
<td>Domestic (DOM)</td>
</tr>
<tr>
<td>Cat</td>
<td>57</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Poultry</td>
<td>65</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cow</td>
<td>112</td>
<td></td>
<td>Livestock (LIVE)</td>
</tr>
<tr>
<td>Horse</td>
<td>136</td>
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<td></td>
</tr>
<tr>
<td>Goat</td>
<td>27</td>
<td>Non-Human</td>
<td></td>
</tr>
<tr>
<td>Seagull</td>
<td>38</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Duck</td>
<td>60</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Possum</td>
<td>28</td>
<td></td>
<td>Wild (WILD)</td>
</tr>
<tr>
<td>Kangaroo</td>
<td>51</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wallaby</td>
<td>57</td>
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<tr>
<td>Koala</td>
<td>7</td>
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</table>
### Average Rates of Correct Classification (ARCC) of known source isolates

Table D.3: ARCC for source isolates categorised as Human versus Non-human determined through Discriminant Analysis

<table>
<thead>
<tr>
<th>Source</th>
<th>Number &amp; %CC isolates classified as</th>
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</thead>
<tbody>
<tr>
<td></td>
<td>Non-Human</td>
</tr>
<tr>
<td>Non-Human (n = 557)</td>
<td>544</td>
</tr>
<tr>
<td>Human (n = 160)</td>
<td>16</td>
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<tr>
<td><strong>Average Rate Correct Class. (ARCC)</strong></td>
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</table>

Table D.4: ARCC for source isolates categorised as Domestic, Livestock, Wild or Human determined through Discriminant Analysis

<table>
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<tr>
<th>Source</th>
<th>Number &amp; %CC isolates classified as</th>
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</thead>
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<tr>
<td></td>
<td>Domestic</td>
</tr>
<tr>
<td>Domestic (n = 179)</td>
<td>141</td>
</tr>
<tr>
<td>Livestock (n = 190)</td>
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<tr>
<td>Wild (n = 188)</td>
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<tr>
<td>Human (n = 160)</td>
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<tr>
<td><strong>Average Rate Correct Class. (ARCC)</strong></td>
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</table>

Figure D.1: Discriminant grouping of Human versus Non-human source isolates
Figure D.1: Discriminant grouping of Human, Livestock, Domestic and Wild source isolates
Table D.5: Classification of unknown source isolates sampled from monitoring locations

<table>
<thead>
<tr>
<th>Monitoring Site</th>
<th>No. Isolates</th>
<th>Source Identification (%) of unknown source isolates</th>
<th>Human</th>
<th>Non-human</th>
<th>Human</th>
<th>Domestic</th>
<th>Livestock</th>
<th>Wild</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cabbage Tree Point</td>
<td>(n = 86)</td>
<td></td>
<td></td>
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</tr>
<tr>
<td><strong>Groundwater</strong></td>
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<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
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<td>CT1</td>
<td>33.6</td>
<td>66.4</td>
<td>33.6</td>
<td>25.4</td>
<td>3.2</td>
<td>37.8</td>
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<td></td>
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<tr>
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<td>49.6</td>
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<tr>
<td><strong>Surface Water</strong></td>
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<td><strong>Surface Water</strong></td>
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</tr>
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<td>66.7</td>
<td>13.3</td>
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</table>
Table D.5: Classification of unknown source isolates sampled from monitoring locations (continued)

<table>
<thead>
<tr>
<th>Monitoring Site</th>
<th>No. Isolates</th>
<th>Source Identification (%) of unknown source isolates</th>
</tr>
</thead>
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<tr>
<td></td>
<td></td>
<td>Human</td>
</tr>
<tr>
<td>Bonogin Valley Groundwater (n = 288)</td>
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<td>50.0</td>
</tr>
<tr>
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<td>94.4</td>
</tr>
<tr>
<td>BO4</td>
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<td>BO5</td>
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<td>36.8</td>
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<td>Surface Water</td>
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<td>60.0</td>
</tr>
<tr>
<td>BOS2</td>
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<td>45.5</td>
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