On-site Wastewater Treatment:

Understanding the Potential Nutrient Risk to Queensland Coastal Waters

A critical review of research literature

Queensland Government

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Executive Summary

In addition to well-documented human health risks associated with enteric pathogens, on-site wastewater treatment systems have recently captured attention as a potential source of nutrients to the environment. Poorly treated wastewater from on-site wastewater treatment systems (OWTS) can reach groundwater and surface waters affecting their environmental values.

This state-of-the-art review critically evaluated published literature on on-site wastewater treatment. It brings together current knowledge on the environmental impacts and related environmental value depletion associated with nutrients and pathogens from OWTS to understand the extent of the potential risk associated with OWTS. The interacting processes between percolating wastewater and soil, and factors influencing those processes are critically reviewed to understand the effectiveness of the treatment occurring in a soil absorption system. The report also reviews the OWTS risk assessment approaches and identifies tools/models that have the potential to be used in the process.

Total nitrogen (TN) in wastewater is removed mainly by biochemical processes, including but not limited to nitrification and denitrification, whereas total phosphorus (TP) is generally removed by physicochemical processes. These processes are sensitive to ambient environmental conditions. Therefore, changes in subsurface conditions such as soil chemistry, pH and texture, oxygen availability and hydraulic conductivity can significantly impact the nutrient removal processes in the subsurface. In addition, site characteristics such as rainfall and temperature, and wastewater characteristics can influence nutrient removal in OWTS. Therefore, this makes OWTS contaminant behaviour and removal highly location specific. Unfortunately, current regulations on design parameters, including buffer distances, are primarily based on preventing disease outbreaks. The effectiveness of the recommended design parameters in preventing nutrient contamination is yet to be investigated.

Most OWTS involve only primary treatment before the release of partially treated wastewater to a confined soil absorption system (SAS). Consequently, it is expected that the majority of nutrient removal would take place in the soil adsorption system and sub-surface soil immediately below the disposal area. The sustainability of nutrient removal depends on a range of factors such as loading rates, climate and soil types, but could be limited. Numerous studies are available from around the world on nutrient impacts on groundwater and surface waters. However, only limited studies have been undertaken in Queensland and Australia.

This critical review has further identified the important knowledge gaps essential for effective risk assessment that currently exist concerning the behaviour and environmental impacts associated with OWTS-sourced nutrients and has provided recommendations for future research. Specific attention was given to identifying knowledge gaps that constrain an in-depth understanding of TN dispersal from OWTS. Only a brief discussion is provided on TP and pathogens.

A growing body of evidence from many parts of the world suggests OWTS as one of the primary sources of nutrients to the environment, particularly TN migration to groundwater, and potentially then to surface water. Despite being one of the main wastewater treatment options in Queensland, knowledge of the wastewater treatment process in OWTS is sparse. This also includes only a limited understanding of the treatment processes that wastewater undergoes in the SAS and the subsurface beneath the treatment system, their efficacy and long-term sustainability in nutrient removal. Studies in other countries also suggested that local conditions such as climate as well as various soil characteristics, including pH, cation exchange capacity, and hydraulic conductivity, can influence wastewater treatment in OWTS. The influence of Queensland conditions on wastewater treatment is yet to be investigated in-depth. The chemical characteristics of the partially treated wastewater discharged into the environment via the disposal area is another factor most overseas studies have identified as being critical for nutrient removal. The influence exerted by these characteristics can vary with the wastewater source and the type of primary treatment in the treatment chamber. For example, ammonia may be predominant in the effluent from a system that provides anaerobic treatment. Further, advanced treatment technologies can significantly influence the quality of the partially treated effluent discharged to the disposal area. The influence of these factors on wastewater treatment processes in Queensland OWTS has yet to be investigated.

Even with the current efforts to limit nutrient inputs to waterways to protect their environmental values, specifically the Great Barrier Reef (GBR), there is a dearth of monitoring studies undertaken to quantify nutrients, particularly nitrate inputs or source tracking to estimate OWTS contribution to total nutrient inputs to ground and surface waters from Queensland catchments. Further, there do not appear to be any case studies which have assessed

the cumulative nutrient loading from OWTS and their impacts on the Queensland environment. The effectiveness of current guidelines, which mainly consider the prevention of public health outbreaks and are applied without considering environmental conditions in nutrient removal, also requires investigations.

In light of studies showing the potential environmental impacts of OWTS, many authorities, including other states in Australia, have introduced frameworks and regulations for planning, approving, managing and monitoring OWTS. Unfortunately, this is not the case in Queensland. Risk assessment frameworks at either the site scale or regional scale are not available. Modelling tools for undertaking mass balance of nutrients or for assessing environmental risk and guidelines for designing and managing OWTS are yet to be established. Regardless of well-documented failure rates in Australia and other parts of the world, protocols for monitoring OWTS at individual or catchment scale is lacking. In summary, tools to assess nutrient removal from OWTS or to assess risk associated with OWTS are lacking in Queensland.

The accurate assessment of environmental risk is essential for well-informed decision-making on OWTS design and implementation and for managing existing systems. This review highlights the importance of investing in further research to understand OWTS-sourced nutrient dispersal and associated risk. Such knowledge will ensure the sustainable application of OWTS and more importantly, facilitate the risk assessment process which is essential for regulatory agencies for decision-making on the development and future investment in wastewater infrastructure.

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List of Abbreviations

BOD	Biochemical Oxygen Demand
COD	Chemical Oxygen Demand
DO	Dissolved Oxygen
EP	Equivalent Persons
GBR	Great Barrier Reef
EV	Environmental Value
TN	Total Nitrogen
TIN	Total Inorganic Nitrogen
TP	Total Phosphorus
OWTS	On-site Wastewater Treatment Systems
SAS	Soil Absorption System

Chapter 1. Introduction

1.1 Background

On-site wastewater treatment facilities treat domestic wastewater in areas where there is no reticulated sewerage system available. These facilities which can treat, store and dispose of sewage generated from premises may include one of the following:

- A wastewater treatment plant using mechanical, biological or filtration methods to treat waste before discharging the effluent to an above or below-ground disposal system.
- A septic system, which uses biological methods to treat waste before discharging the effluent to a below-ground system; or
- A dry-vault system that employs a chemical, composting or incinerating approach.

On-site wastewater treatment is common in rural and regional areas and the peri-urban fringe of major urban centres in Queensland and Australia. There was significant research activity in the 1970s to early 2000 on the potential human health and environmental impacts arising from poorly designed systems and lack of maintenance by householders. Since that time, there has been a paucity of new research, possibly based on the assumption that the key knowledge gaps have been resolved or the remaining issues do not warrant further investigation.

In earlier days, on-site treatment was meant to treat wastewater to reduce human health impacts. However, the removal of additional wastewater constituents including nutrients, organic matter and other contaminants, namely, organic and inorganic compounds is also considered to be important for protecting human health and environmental values. In recent years, there has been increasing concern among regulatory authorities about the potential adverse impacts arising from poorly treated sewage effluent from on-site wastewater treatment systems (OWTS) containing nutrients, particularly total nitrogen entering surface water and groundwater resources. This has particularly been the case in coastal areas of Queensland, where there is a significant focus on reducing nutrient runoff and inputs into estuaries, bays and particularly the Great Barrier Reef (GBR) catchments. However, the potential for the release of nutrients from OWTS has not been a focus of state or local government investment to date, even though current research would suggest that they are a potential pollutant source.

Small OWTS (below 21 equivalent persons - EP) are regulated under the Plumbing and Drainage Act 2018, but also require local government approval. Unfortunately, it is not easy to estimate the potential risk from unsewered areas involving these small systems given the lack of information about their geographical distribution, the number of such systems in the various jurisdictions and the lack of readily available assessment tools.

Larger sewage treatment systems (greater than 21 EP) require approval under the Environmental Protection Act administered by the Department of Environment, Science and Innovation. Recently, there has been an increase in applications for commercial-scale OWTS in coastal areas that entail the disposal of effluent from the treatment system to groundwater. Again, there are limited tools available to assess the potential risk posed by such systems, and if approved, these could also contribute to nutrient loads being released to coastal areas.

1.2 Research Questions

This review focused on the following priority questions concerning on-site wastewater treatment systems:

1. What are the human and ecosystem health consequences of poorly treated effluent from OWTS entering ground and surface water?
2. What are the critical knowledge gaps that currently exist that constrain informed decision-making concerning the regulation of nutrient releases from OWTS?
3. What tools and techniques are currently available to assess potential risks associated with OWTS?
4. Does adequate knowledge currently exist to assess the potential risk of nutrient releases from OWTS in coastal areas of Queensland at the individual site, suburb or cluster scale?

1.3 Aims and Objectives

This review aimed to conceptually understand nutrient releases from on-site wastewater treatment, primarily total nitrogen, in the context of mobility and processes in the soil environment, that will support informed decision-making.

The primary objectives of this review were:

1. Investigate the influence of the treatment system characteristics and wastewater characteristics in nutrient dispersal from OWTS.
2. Critically review and identify current knowledge gaps in relation to nutrients, particularly total nitrogen (TN) removal in wastewater disposal systems.
3. Identify current knowledge gaps on processes and mechanisms that influence the fate and transport of nutrients, particularly TN, in the subsurface.

1.4 Scope

This review was conducted to determine whether currently available knowledge on on-site wastewater treatment can be used to help guide future investment, regulation and policy development for small and large-scale OWTS in Queensland. The emphasis was on nutrient removal, particularly TN, as impacts from nutrients on aquatic ecosystem environmental values (EVs) are important in the Great Barrier Reef and other catchments in Queensland. The impacts associated with total phosphorus (TP) and pathogens have been discussed briefly. The critical review was based solely on literature available in the public domain. No independent research was undertaken.

1.5 Report Overview

Chapter 1: Introduction

The introduction provides the context for this study including the background and the aims and objectives. The chapter also presents the key research questions aiming to be answered in this state-of-the-art literature review. The overview of the study undertaken presented in the form of a schematic diagram is also included in this chapter.

Chapter 2: Potential Impacts of OWTS

This chapter presents a critical review of the literature relating to human and environmental health impacts from the use of OWTS. The study was not limited to human health impacts but also extended to the ecosystem impacts from contaminants associated with effluent from OWTS. A particular focus was the impact of nutrients from OWTS, especially TN, on fresh and marine waters. Further, available knowledge on the potential deterioration of EVs is also presented in this chapter.

Chapter 3: Factors Influencing Total Nitrogen Dispersal from OWTS

Key findings presented in the literature on TN removal from OWTS are critically discussed in this chapter including processes and mechanisms that influence TN removal. In the absence of key knowledge on the influence of soil characteristics on denitrification, a cross-disciplinary approach was adopted. Knowledge derived from agricultural research was used, with the understanding that the actual impacts of TN originating from OWTS may vary due to the difference in wastewater characteristics and dispersal mechanisms. The chapter also identifies key knowledge gaps that constrain an in-depth understanding of TN dispersal from OWTS.

Chapter 4: Factors Influencing Total Phosphorus Dispersal from OWTS

This chapter presents existing knowledge on TP removal from OWTS. The chapter also identifies key processes that contribute to immobilising TP in the subsurface and the potential for long-term leaching of TP to groundwater as a result of dissolution and desorption. More importantly, the chapter identifies limitations in current knowledge about TP removal from OWTS.

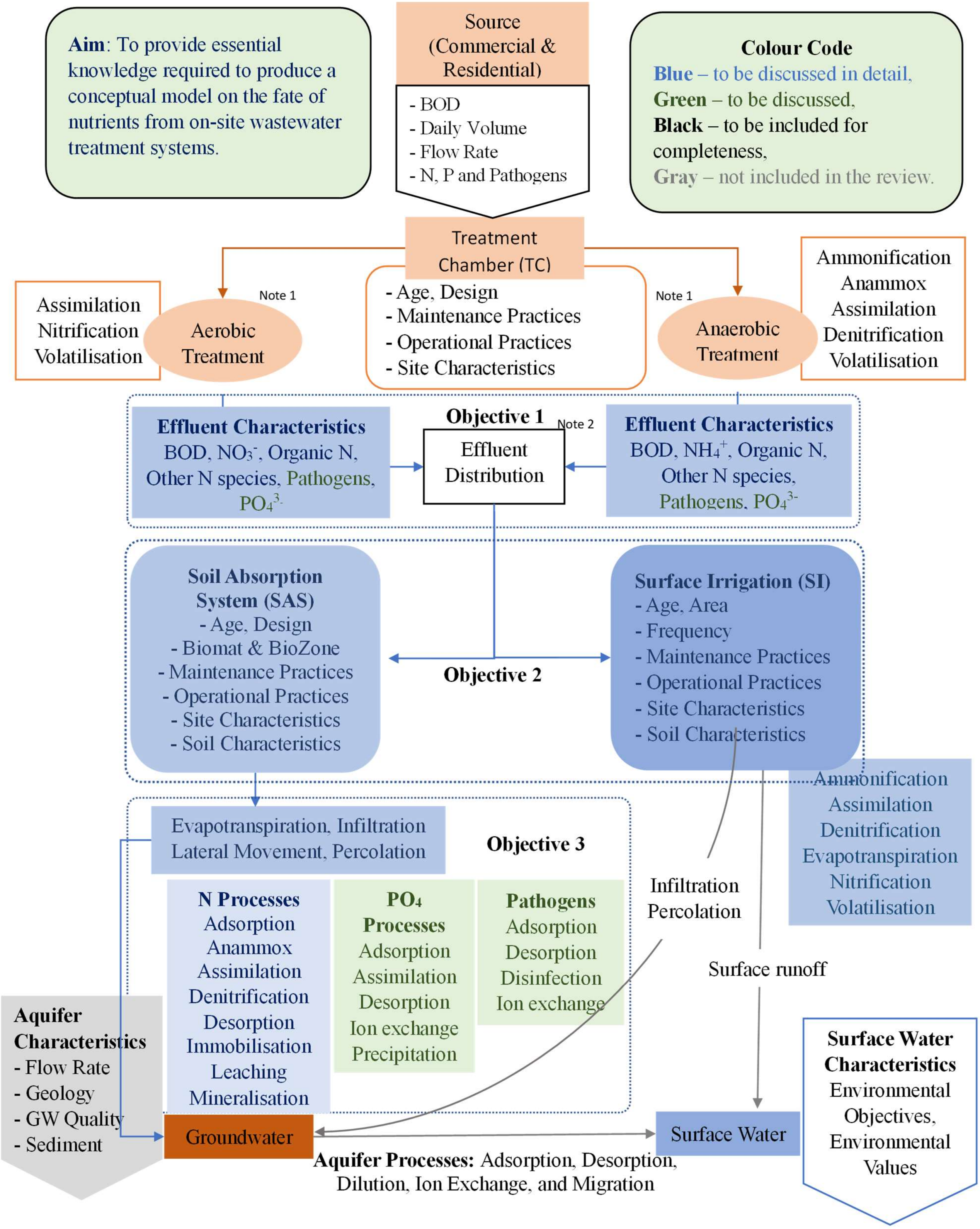
Chapter 5: Evaluation of Nutrient Removal Performance of OWTS

This chapter reviews the currently available tools and techniques for the assessment of risk associated with nutrients, particularly TN, from OWTS. In addition to the risk assessment models developed specifically for OWTS, models that have been developed for other contaminants and have the potential to be used for OWTs are also presented. The chapter introduces a 'failure' definition for OWTS in the absence of a common definition in the literature.

Chapter 6: Conclusions

This chapter summarises the key findings of this state-of-the-art review including the key knowledge gaps that merit future research.

1.6 Conceptual Framework for the Project



Note [1] Range of different configurations exists, also a treatment train could be adopted.

Note [2] Range of devices including perforated pipes and a distribution box could be used.

Chapter 2. Potential Impacts of OWTS

2.1 Introduction

Domestic wastewater contains nutrients, pathogens, organic matter, and suspended solids in significant concentrations (NHMRC, 2011). These constituents need to be removed or mitigated to an acceptable level before releasing to the environment to avoid potential adverse human and ecosystem health impacts. Under favourable operational conditions, on-site wastewater treatment systems (OWTS) could potentially meet these requirements (USEPA, 2002). However, after the disposal of the partially treated effluent to land, a certain amount of wastewater contaminants can percolate and infiltrate into groundwater (Lusk et al., 2017) and subsequently reach surface water during recharge events. This can make OWTS a key nonpoint pollution source of sewage contamination (Ahmed et al., 2005). Figure 2.1 illustrates the potential impacts of OWTS in the environment.

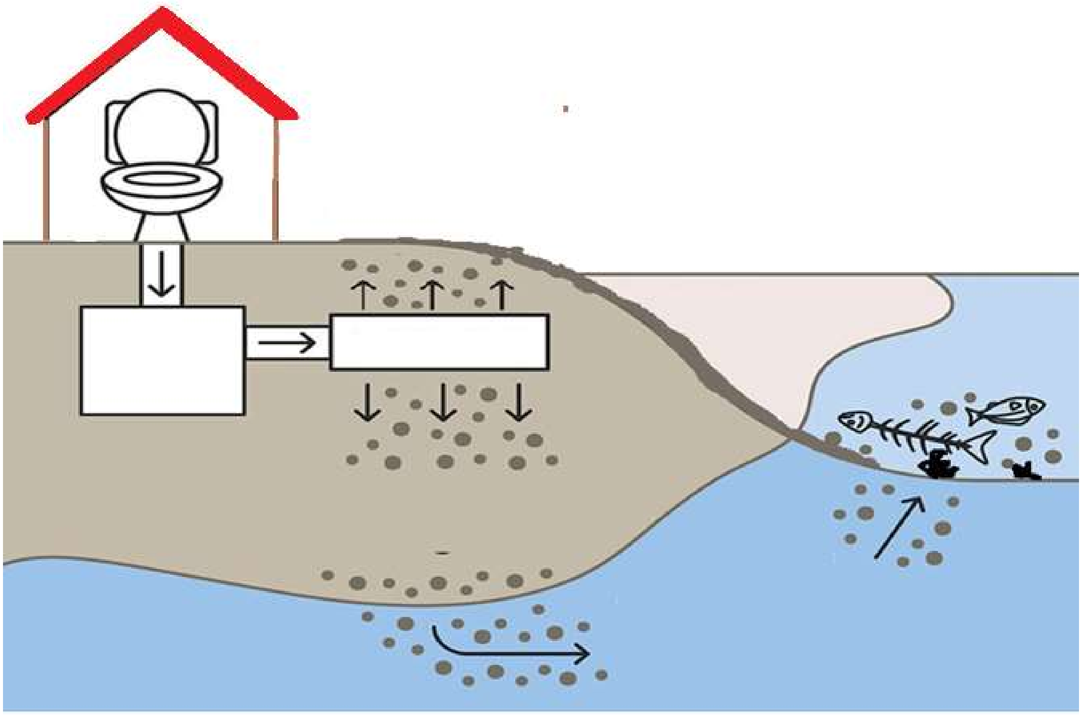


Figure 2.1 Potential impacts of OWTS

Nutrients, including nitrogen and phosphorus, in wastewater can act as individual stressors to the receiving environment (Wear and Thurber, 2015). For example, nutrients can promote the growth of microorganisms in the soil absorption system (primary receiving environment) (Beal et al., 2004). Further, nutrients can pose detrimental impacts on the environmental values of ground and surface waters (secondary receiving environments, Wear and Thurber, 2015). Given the diverse impacts on human and ecosystem health, Gunady et al. (2015) described nutrients as the main hazard associated with OWTS. However, the human and ecosystem health impacts of enteric pathogens from OWTS are also important. Several public health emergencies associated with human enteric pathogens from OWTS have been reported over the years (Borchardt et al., 2011; Pradhan et al., 2008). This chapter discusses the environmental impacts of nutrients from OWTS and the potential decline in environmental values, with a specific emphasis on nitrogen and its various forms. Further, significant knowledge gaps have been identified that constrain effective risk assessment and thus, management. Recommendations are also provided for further research.

2.2 Impacts on Environmental Values and Ecosystem Health

OWTS utilise processes and mechanisms in the natural environment to achieve wastewater treatment. In most cases, dedicated soil absorption systems are used as the final disposal method, where the sorption properties of soils and subsurface biochemical processes such as assimilation and denitrification are relied on for further mitigation. In general, subsurface wastewater disposal methods are not capable of treating wastewater to an acceptable level to avoid detrimental environmental impacts due to several reasons, including but not limited to poor design and maintenance practices, siting and incompatible soil types such as sandy soils in coastal locations.

Crop irrigation is another common approach used to disperse the effluent being discharged from the treatment chamber. In this case, plant uptake contributes relatively more to the treatment process followed by soil treatment processes (Tennakoon & Ramsay, 2020). This is the preferred approach for commercial-sized on-site treatment in Queensland. Readers are referred to the publication, 'Disposal of Effluent using Irrigation, Technical Guideline' by Tennakoon and Ramsay (2020) for further details.

Wastewater contaminants from OWTS can reach ground and surface waters due to a range of factors including, hydraulic failure resulting from extensive loading, extensive clogging or exhaustion of the soil absorption system, poor maintenance practices and limited nitrification-denitrification processes in the subsurface. Contaminants can adversely affect the quality of the environment and impact its uses, which are defined as 'Environmental values' (EVs) under the Australian Water Quality Guidelines (NHMRC, 2011). Environmental values for ground and surface waters include:

- aquatic ecosystem health
- aquaculture and human consumption of aquatic foods
- agricultural uses
- recreational uses
- drinking water
- industrial uses
- cultural and spiritual values
- aesthetics

The potential ramifications of contaminants released from OWTS to the natural environment and the associated adverse impacts on environmental values are discussed below.

2.2.1 Impacts on water quality and degradation of ecosystem values of freshwater

a) Groundwater impacts

Groundwater is susceptible to microbial and nutrient contamination from OWTS effluent (Carroll & Goonetilleke, 2005; Goonetilleke et al., 2007). Effluent from the treatment chamber is only partially treated and contains a range of nitrogen forms, including organic nitrogen and ammonia (NH_3) in addition to other contaminants. In soil absorption systems, organic nitrogen and ammonia can be nitrified before being attenuated by denitrification. Other processes such as assimilation, adsorption, and plant uptake can also immobilise nitrogen species in wastewater (Robertson et al., 2012; USEPA, 2002). Depending on a suite of factors including site, soil and wastewater characteristics, these processes can reduce organic nitrogen, ammonia and nitrates (NO_3^-) present in a soil absorption system. A detailed discussion of the key factors influencing nitrogen dispersal in soil absorption systems is provided in Chapter 3 of this report.

However, an appreciable amount of total nitrogen (TN; nitrate, nitrite, ammonium and organic nitrogen) can leach into the groundwater if there is limited attenuation in the soil absorption system. Summarised in Table 2.1 are a selection of key studies that have investigated TN loading from OWTS to groundwater in different regions of the USA. For example, De and Toor (2016) estimated that the TN influx to shallow groundwater from 2.5 million OWTS in Florida can amount to 2.4×10^6 kg/year. Oliver et al. (2014) found higher TN concentration in the groundwater in Gwinnett County, Georgia, downgradient from a soil absorption system compared to the average groundwater. William et al. (2007) estimated TN loading from OWTS into shallow groundwater in the Wekiva region, Florida as 1.2 – 5.5 kg/capita/year. Lapointe (1990) compared year-long groundwater monitoring data in canal residences with OWTS in the Florida Keys with a control site in the National Key Deer Wildlife Refuge. That study reported 400 times higher total inorganic nitrogen (TIN; nitrate, nitrite, and ammonia) concentration in the OWTS-impacted groundwater compared to groundwater at the control sites. The findings from the various studies cited above clearly indicate that OWTS has limited ability to attenuate TN and the groundwater in regions with OWTS is susceptible to nitrogen contamination. Based on the current review, it is evident that no comprehensive studies have been conducted in Australia to determine TN input from OWTS into groundwater.

Table 2.1 OWTS-originated TN inputs to groundwater (studies undertaken in the USA)

Findings	References
2.4×10^6 kg/year TN loading to shallow groundwater from OWTS in Florida.	De and Toor (2016)
Above average TN concentration in groundwater downgradient from the soil absorption system.	Oliver et al. (2014)
3.0 kg NO_3-N /capita/year loading from OWTS to waters in the Wekiva Basin, USA.	MACTEC (2010)
1.2 – 5.5 kg/capita/year TN loading to shallow groundwater in the Wekiva region, Florida.	William et al. (2007)
400 times higher TN concentration in OWTS-impacted groundwater compared to groundwater in control areas.	Lapointe (1990)

In general, TP may be effectively removed by the soil absorption system. Therefore, groundwater contamination from OWTS-sourced TP is often less common. On occasion, groundwater TP concentrations greater than background values have been reported in past studies (Efroymson et al., 2007). This is particularly attributed to poor OWTS maintenance practices (Lusk et al., 2017). In groundwater, TP is readily adsorbed to sediments. TP attenuation in the subsurface is discussed in detail in Chapter 4 of this report.

Pathogens from OWTS can also degrade groundwater quality. Yates and Yates (1989) concluded that treatment chamber effluent is the most significant source of pathogenic bacteria and viruses in the subsurface. In groundwater, pathogens can remain effective and move rapidly tens to hundreds of meters within weeks (Borchardt et al., 2007; Bradbury et al., 2013; Scandura & Sobsey, 1997) contaminating waterbodies far from the source. The degradation of water quality due to pathogenic contamination can adversely impact the environmental values of groundwater, particularly resources used for drinking water, recreation and groundwater-dependent ecosystems. Public health outbreaks resulting from groundwater contaminated with treatment chamber effluent have been reported widely as discussed in Section 2.3.

OWTS can contribute to groundwater recharge. Oliver et al. (2014) modelled streamflow with and without OWTS at Gwinnett County, Georgia, USA, using the Soil and Water Assessment Tool (SWAT) catchment-scale model. Water balance model simulations with and without the presence of OWTS estimated a water yield increase of 3.1% at the catchment scale and 5.9% in areas with high OWTS density. Although the increase in water yield was relatively low, considering the overall extent of land area served by OWTS (0.88 – 1.62% of total land), the contribution to groundwater recharge can be considered to be significant.

The environmental values of groundwater as a drinking water source can be adversely impacted due to OWTS. An example cited in the literature is Orange County, USA (MACTEC, 2010), where OWTS-originated nitrate concentration in groundwater exceeded the WHO standard for drinking water (50 mg/L, WHO 2017). In addition, groundwater connected to surface waters can have adverse impacts on surface water environmental values such as the protection of aquatic ecosystems, recreational values, and industrial uses. The potential impacts on surface water environmental values in the context of OWTS contaminant release are discussed below.

b) Impacts on surface freshwater systems

Nutrient availability in freshwater environments influences their primary production (Mueller & Helsel, 1996). OWTS can significantly contribute to nutrient enrichment in surface waters (Table 2.2). Hunter and Walton (2008) estimated an influx of 97 TN kg/ha/year to the Johnstone River, Queensland, from OWTS-served residential areas in the catchment which was two times greater than that from sugar cane and banana cultivated areas in the catchment (38 and 42 kg N/ha/year, respectively). OWTS was found to contribute 11% of the total annual TN influx into the catchment if the contribution of other sources such as animal waste and household-level fertiliser application in unsewered regions was assumed as not significant. Similar studies in other catchments are required to estimate cumulative TN loading from OWTS to the Queensland waters. Further, comprehensive studies utilising source tracking techniques will be able to provide accurate estimations of TN loading.

In Western Australia, an investigation was conducted to estimate the nutrient loading from different land use types in the Peel-Harvey catchment. The TN influx from OWTS was calculated using per-person septic tank emission rates estimated by Whelan and Barrow (1984), the number of reticulated sewerage systems and average occupancy rates. According to the study, OWTS was attributed to 3.7% of the TN loading to the Peel-Harvey catchment (2.3×10^4 TN kg/year, Hennig et al. 2021). This value reflects the direct influence of OWTS based on inputs from individual systems. OWTS can also contribute to TP loading to surface waters as listed in Table 2.2. Based on the extensive review of the literature undertaken, it is evident that comprehensive studies have not been conducted in Queensland to determine the nutrient loading and impacts of OWTS to surface waters.

Table 2.2 OWTS-originated nutrient loading to surface waters in Australia

OWTS contribution	References
Approximately 10.8 g TN /capita/year (11% of TN) loading to the Johnstone River, Queensland	Hunter and Walton (2008)
3.7% TN loading (2.3×10^4 kg/year) in the Peel-Harvey catchment, Western Australia	Hennig et al. (2021)
2.6% TP loading (1.6×10^3 kg/year) in the Peel-Harvey catchment, Western Australia	Hennig et al. (2021)

Excess nutrients can influence the trophic status of surface waterbodies by inducing the growth of weeds, cyanobacteria, and other toxic algae (eutrophication). An example cited in the literature is the Peel-Harvey catchment in Western Australia where OWTS-derived nutrients were identified as a potential cause of frequent eutrophication events (DWWA, 2010; Williams, 2009). Eutrophication and related habitat degradation can significantly impact aquatic ecosystem health (Figure 2.2). It can also diminish the EVs of surface waters as a source of drinking and industrial water. The degraded aesthetic appearance and function of surface water due to eutrophication can impact recreation-based tourism. The trophic state of surface waters can also impact their spiritual value. Comprehensive research does not appear to have been undertaken to date in the context of attribution of OWTS impacts to EVs of surface waters in Queensland. This results in a significant knowledge gap which merits future research.

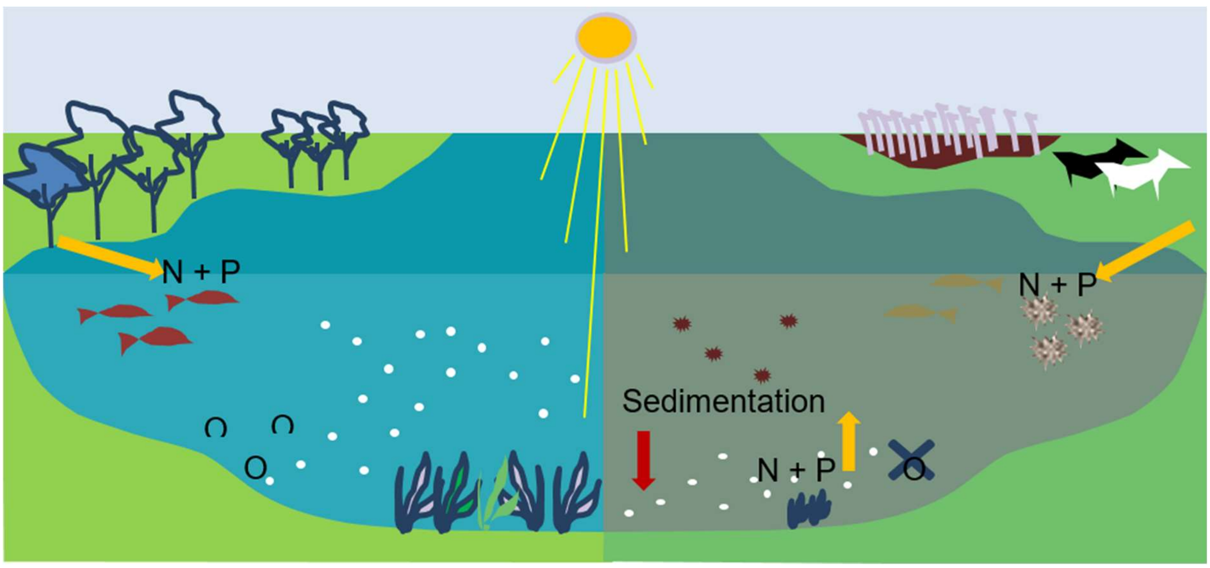


Figure 2.2 Illustration of the depletion of ecosystem values resulting from OWTS nutrient loading

OWTS can be a significant source of pathogens to surface waters. Carroll and Goonetilleke (2005) observed a relatively high percentage of human *E.coli* isolates in OWTS-served areas in the Gold Coast region, Queensland. According to the NHMRC (2011), septic tanks located on beaches pose a high risk to human health arising from swimming and potential seawater contamination with microbial pathogens. Recreation in marine and fresh waters contaminated with human pathogens from OWTS can result in a range of health impacts such as skin and eye infections as well as gastrointestinal diseases (Tillett et al., 1998). Human health impacts on drinking and recreation waters contaminated with human enteric pathogens are discussed in greater detail in Section 2.3.2. The role of human enteric pathogens in coral diseases is discussed in Section 2.2.2 and Appendix A. Loss of coral and increased pathogen concentrations in recreational water may lead to lower visitation triggering economic impacts on tourist attractions such as the coastline of Queensland. Limited available literature (Carroll & Goonetilleke, 2005; NHMRC, 2011; Tillett et al., 1998) indicates that OWTS-originated pathogens can adversely impact the environmental values of surface waters.

2.2.2 OWTS impacts on water quality and degradation of marine ecosystem values

a) Impact of nutrients

Effluent discharge from the treatment chamber can result in significant impacts on coastal ecosystems as well as water-based recreation (Cordero et al., 2012). Nutrient enrichment of coastal ecosystems is a major concern in freshwater, estuarine and marine water. This can trigger ecosystem changes, including harmful algal blooms, hypoxia, fish kills, and loss of biodiversity, as well as the degradation of water quality (Conley et al., 2009; Diaz & Rosenberg, 2008; Paerl, 2009; Withers et al., 2011).

The research literature is replete with numerous case studies where wastewater-borne TN from catchments and point sources has been identified as the main contributor to the eutrophication of coastal waters, such as the Indian River Lagoon in Florida, Waquoit Bay in Massachusetts, USA and the Baltic Sea, Sweden (Boesch et al., 2006; Kroeger et al., 2006; Lapointe et al., 2015). Dissolved inorganic and organic nitrogen can migrate into coastal areas during base-flow conditions or streams recharged by groundwater during the low-flow period (Hunter & Walton, 2008; Oliver et al., 2014). The marine environment is generally oligotrophic, and anthropogenic nutrient inputs can significantly impact its ecological health. For example, TIN has been found to cause excessive macroalgae biomass across the Southeast Florida Shelf (Lapointe et al., 2005). Macroalgae blooms can overgrow seagrasses and adult corals, obstruct juvenile coral development and develop areas of anoxia and hypoxia, depleting fish populations and affecting biological diversity (Lapointe et al., 2005).

In Australia, Bell (1991) identified sewage discharges as a potential source of total phosphorus (TP) and phytoplankton levels above eutrophication in coral reef water in the Great Barrier Reef (GBR). There are 36 seagrass community types defined by their species assemblages recorded in the Great Barrier Reef World Heritage area (Carter et al., 2021). Eutrophication can threaten these communities which provide habitat and food sources for the endangered dugong and turtle populations. Similarly, the impact on seagrass populations can also threaten the other ecosystem services that seagrass provides, such as coastal protection, bacteria removal, sediment particle trapping, and acting as a carbon sink. A detailed discussion of the impacts on coral health due to increased nutrient concentrations in ambient waters is provided in Appendix A. As reports are rare, impacts on the GBR ecosystem due to increased nutrient and pathogen influx from OWTS particularly with the increasing human population and booming tourism industry in the region need to be investigated (https://environment.des.qld.gov.au/management/water/water-quality-improvement-plans).

b) Pathogens and coral

The impact of wastewater pathogens on coral health is largely an under-explored area. Only a limited number of studies have investigated the link between human enteric pathogens and coral diseases. White pox disease causing *Serratia marcescens* is one example. Wastewater has been identified as the potential source of *Serratia marcescens* in the coral reefs in the Caribbean and Florida Keys (Patterson et al., 2002; Sutherland et al., 2010; Sutherland et al., 2011). Figure 2.3 shows *A. palmata* colony loss between 2008 and 2009 due to white pox disease caused by the human opportunistic pathogen *Serratia marcescens* at Looe Key, Florida Keys. Abdelzaher et al. (2010) confirmed this finding using faecal indicator bacteria and human-associated microbial and source tracking (MST) markers. Their study hypothesized that waterways carrying nonpoint source sewage pollutants as the cause of this contamination. Darden (2000) identified a positive correlation between coral diseases and increased bacteria levels. The white-band disease is another disease commonly associated with human enteric pathogens. This disease caused 95% of *Acroporid species, Acroporid palmitate,* and *Acroporid cervicornis* coral colonies to be lost in the Caribbean (Aronson & Precht, 2001; Miller et al., 2002). In the GBR, seven coral disease episodes, including protozoan, and *Vibrio spp*. infections have been documented (Harvell et al., 2007). However, their link to OWTS has not been established nor any studies undertaken in Queensland could be identified that have related OWTS to coral diseases. As such, this constitutes an important knowledge gap and an area for future research.

Figure 2.3 Images of coral (*A. palmata* colony) growth and partial mortality due to white pox disease at Looe Key, Florida Keys in June 2008, June 2009 and July 2009. Note: The 3 cm diameter scale bars can be seen in the image (Sutherland et al., 2011)

In addition, the growth of these pathogens can be induced by nutrient inputs from OWTS (Kuntz et al., 2005; Vega Thurber et al., 2014). Pathogens, including bacteria and viruses, can reduce the overall clarity of the water which may prevent sunlight from reaching the corals. This is detrimental to the photosynthetic algae cells which are symbiotic with corals. However, the relationship between coral health and reduced clarity (turbidity) of ambient water due to OWTS-source pathogens is yet to be investigated in-depth.

2.2.3 Effects on soil quality due to OWTS

Wastewater treatment in the soil absorption system involves dynamic interactions between the physical, chemical and biological processes (McKinley & Siegrist, 2011). These interactions can cause changes in soil characteristics. Immediate changes to the soil microbiome are visible under the wastewater disposal system just after the start of the operation. Nutrients in partially treated wastewater will induce microbial growth resulting in the production and accumulation of cells and organic by-products. These microbes and their byproducts help form a biomat, which is beneficial in terms of providing effective treatment of wastewater by acting as a filter and regulating the effluent flow into the subsurface. Further, the biomat directly influences the wastewater treatment process by reducing organic matter, nutrients, and pathogens input. It provides a steady hydraulic conductivity rate which ensures the unsaturated conditions needed for the nitrification or biological oxidation of wastewater ammonia-nitrogen (Beal et al., 2006; USEPA, 2002). However, nutrients can pass through the biomat resulting in biological and chemical changes in a soil absorption system. A summary of the physical, biological, and chemical changes that can occur in the soil absorption system due to wastewater application is detailed in Table 2.3.

Only a limited number of studies have investigated the impact of OWTS on soil characteristics. Beal et al. (2006) and Beal et al. (2005) investigated biomat formation from wastewater application to the soil absorption system. Dawes (2005) investigated the changes in soil characteristics resulting from wastewater disposal. The age and density of OWTS in an area can also influence soil characteristics. For example, with the ageing of OWTS, significant biozone development and TP accumulation are possibilities. The impacts of a high OWTS density may include high microbial activity in the subsurface as well as a rising groundwater table. However, the influence of age and density of OWTS on changes to soil characteristics is yet to be fully understood.

Table 2.3 Some common impacts of subsurface wastewater disposal on soil characteristics

Soil Characteristic	Impact	References
Soil microbiome	Formation of high microbial active biozone.	Burford and Bremner (1975)
	growth of denitrifying bacteria growth of anammox bacteria under oxygen-limiting conditions	Robertson et al. (2012); Wijaya et al. (2017)
Hydraulic conductivity	Decreases with time due to sodium build-up	Dawes (2005)
Soil pH	1 – 2 point reduction due to nitrification and organic matter decomposition	Eveborn et al. (2014); Robertson (2003)
Redox conditions	Above the biomat - anoxic conditions Below the biomat - oxidising conditions	Beal et al. (2006); Beal et al. (2004)
Soil Chemistry	Dissolution of carbonate minerals	Zanini et al. (1998)
	Precipitation of phosphorus with $Fe^{2+/3+}$, Al^{3+} and Ca^{2+}	

2.2.4 Air quality impacts due to OWTS

OWTS contributes to global warming by emitting greenhouse gasses, namely, N_2O, CH_4 and CO_2 (Diaz-Valbuena et al., 2011; Truhlar et al., 2016; USEPA, 2010). Table 2.4 provides a summary of OWTS greenhouse gas emissions. The CO_2 emissions from OWTS vary between 0.07 – 0.25 tonnes CO_2 eq/capita/year. This is considerably higher than the average per capita CO_2 emissions in Australia in 2016 (0.048 tonnes CO_2 eq/capita/year, https://www.worldometers.info/co2-emissions/co2-emissions-per-capita/). Therefore, OWTS can be considered a significant source of greenhouse gases. Besides global warming, CH_4 from OWTS can affect the amenity of the surrounding environment due to offensive odour (Gunady et al., 2015).

OWTS systems also emit several gasses other than greenhouse gases. This includes NH3 and N2 resulting from the mineralisation and denitrification processes in OWTS. To date, no research appears to have been undertaken to quantify the emission of these gasses from OWTS. Investigations into net nitrogen gas emissions are important for determining the nitrogen mass balance of OWTS.

Table 2.4 Greenhouse gas emissions from OWTS

Reference	OWTS type/ feature	CH_4 est. ($g\ CH_4$/ capita/day)	CO_2 est. ($g\ CO_2$/ capita/day)	N_2O est. ($g\ N_2O$/ capita/day)	Anthropogenic emissions (tonnes CO_2 eq/capita/year)	Biogenic emissions (tonnes CO_2 eq/capita/year)
Winneberger (1984)	Septic Tank	8-11	5 – 6	nr	0.07-0.10	1.8×10^{-3} –2.2×10^{-3}
IPCC (2007)	Septic Tank	25.5	cb	nr	0.23	nr
USEPA (2010)	Septic Tank	27.1	cb	nr	0.25	nr
Diaz-Valbuena et al. (2011)	Flux chamber	11.0	33.3	0.005	0.10	0.012
Diaz-Valbuena et al. (2011)	Septic Tank	10.7	335	0.2	0.12	0.12
Truhlar et al. (2016)	Septic Tank	11	160	0.12	nr	nr
Truhlar et al. (2016)	Sand Filter	0.0072	120	0.0060	nr	nr
Truhlar et al. (2016)	Drain Field	-0.0038	130	0.022	nr	nr

est.- estimated
cb - emissions considered biogenic in wastewater treatment
nr – not reported or not available

2.3 Human Health Impacts from OWTS Effluent

Major human health impacts of OWTS releases are associated with nutrients and enteric pathogens. Consumption of ground and surface water, as well as primary recreation in surface waters bodies contaminated with pathogens, can result in waterborne disease outbreaks. The discussion below provides an overview of the potential human health impacts arising from the release of OWTS contaminants.

2.3.1 Impacts of nutrients on human health

OWTS can contaminate drinking water with nutrients including nitrite (NO_2^-), nitrate (NO_3^-), ammonia (NH_3) and phosphates (PO_4^{3-}). Ingestion of formula made from drinking water contaminated with nitrate (>50 mg/L) is identified as the cause of methemoglobinemia, also known as blue baby syndrome (WHO, 2017). However, according to available studies, there is only a poor relationship between methemoglobinemia and OWTS (L'Hirondel & L'Hirondel, 2002). There are several sources other than nitrate from OWTS that can cause methemoglobinemia. These include chemical compounds such as benzocaine and nitrobenzene, antibiotics including dapsone and chloroquine and nitrites used to preserve meat from spoiling (Mansouri & Lurie, 1993). Table 2.5 summarises nitrate-related human health effects reported in the research literature.

Table 2.5 Selected human health impacts of nitrates discussed in the research literature

Disease	Details	References
Methemoglobinemia	Fatal for infants younger than three months. Pregnant women and people with a deficiency of glucose-6-phosphate dehydrogenase or methemoglobinemia are susceptible.	NHMRC (2011); WHO (2017)
Carcinogenic impacts	Non-Hodgkin's lymphoma, Prostate cancer	Aschebrook-Kilfoy et al. (2012); Wu et al. (2013)
Thyroid gland effects	Inhibits thyroidal iodide uptake	WHO (2016)
Reproductive and developmental toxicity	Birth defects in offspring.	Huber et al. (2013); Brender et al. (2013)

To the best of available knowledge, no human health effects have been reported relating to other OWTS-originated nutrients including ammonia and phosphate in drinking water. However, water contaminated with ammonia may not be suitable for drinking as it may indicate sewage contamination or microbial activity (NHMRC, 2011). In addition, nitrate and phosphate can indirectly impact human health by inducing the growth of cyanobacteria (blue-green algae), which produces harmful cyanotoxins.

2.3.2 Impact of enteric pathogens on human health

Human faeces contain enteric pathogenic microorganisms including bacteria, viruses, protozoa and helminths (Scott et al., 2002). The most common human health impact associated with the consumption of water contaminated with human pathogens is enteric illnesses. One example is gastrointestinal illness (diarrhoea) which results in 1.7 billion cases annually and remains the second leading cause of death in children under five years of age (Efstratiou et al., 2017; Kotloff et al., 2013). Appendix B provides a summary of waterborne diseases associated with human enteric pathogens.

In general, under optimal operational conditions, well-designed and maintained OWTS can inactivate pathogens adequately to help avoid a public health outbreak (Katz & Rosenbaum, 2010). Adsorption, straining, desiccation, and predation by soil microorganisms are the main mechanisms that attenuate pathogens originating from OWTS. Adequate lateral and vertical setback distances can prevent OWTS-originated pathogens from entering surface and groundwater resources (USEPA, 2010). However, OWTS has been identified as the source of pathogens in several outbreaks as listed in Table 2.6. This indicates that setback distances do not necessarily ensure adequate pathogen removal from OWTS. Proper design practices, particularly in problematic soils can prevent unfortunate events such as the norovirus cases in Northeastern Wisconsin reported by Borchardt et al. (2011). Similarly, good maintenance practices could have avoided contamination such as high faecal coliform concentrations in Wallis Lake, New South Wales, Australia (Conaty et al., 2000).

Other reasons for poor pathogen removal include the influence of subsurface characteristics such as the degree of saturation, temperature, soil type, soil structure, and depth of groundwater (Arnade, 1999; Bloetscher & Van Cott, 1999; Dillon et al., 1999; Lusk et al., 2017). Another reason could be the robustness of pathogens. Some pathogens can survive days, if not months in moist, neutral soils (Lusk et al., 2017). Removing pathogens, particularly viruses and protozoa after entering drinking water sources is a challenge. This is because viruses and protozoa are resistant to common disinfection methods (King and Monis, 2007; Ryan et al., 2016; USEPA, 2002).

Therefore, it is essential to take all possible measures to prevent water sources from being contaminated with pathogens from OWTS in the first instance.

Table 2.6 Selected list of public health outbreaks associated with OWTS

Outbreak	Source/contamination	References
Norovirus GII outbreak (17 Cases)	Contaminated drinking water	Ryan et al. (2016)
447 Hepatitis A cases (1997) in Wallis Lake in NSW	Contaminated oysters	Conaty et al. (2000); Pradhan et al. (2008)
229 norovirus cases in northeastern Wisconsin (2007)	Contaminated groundwater	Borchardt et al. (2011)
Chattahoochee River National Recreation Area, Metropolitan Atlanta, Georgia, 1994 -1995	Reduced the recreational value of the river	Gregory and Frick (2000)

Bai et al. (2020) reported on the genotoxicity associated with OWTS effluent. In an investigation conducted using the UMU test, the International Organization for Standardization standard test (ISO13829, 2000) for evaluating the genotoxicity potential, the study reported the ability of treatment chamber effluent to damage salmonella DNA, indicating genotoxicity. The exact causality of genotoxicity and its impact on human health is yet to be determined. However, this highlights the importance of further research into the diverse health impacts associated with OWTS effluent. Factors influencing pathogen removal in soil absorption systems are given in Appendix C of this report.

2.4 Summary of Key Findings

OWTS largely use subsurface disposal to achieve adequate treatment of the partially treated wastewater being discharged from the treatment chamber. Subsurface wastewater disposal systems use the sorption capacity and biochemical processes of the soil for further treatment. In comparison, surface wastewater disposal systems (land irrigation) employ plant uptake in addition to soil sorption capacity to treat the wastewater. This review has shown that OWTS using subsurface disposal can contaminate groundwater and surface water with nutrients and pathogens. These contaminations can result in significant adverse impacts on the environmental values of both surface and groundwaters including drinking water, aquatic ecosystems, and recreational water. In addition to surface and groundwater contamination and consequent environmental value impacts, OWTS can also impact the environmental values of air and soil.

OWTS can be considered a substantial source of nutrients, particularly TN inputs to groundwater. Although soil has a high capacity to adsorb phosphorus, the risk of groundwater contamination by OWTS-originated phosphorus is also inevitable. However, investigations into OWTS as a potential source of nutrients in Australian groundwaters are limited. The only OWTS-related nutrient influx study in Queensland that could be identified was based on the Johnston River catchment. That study concluded that OWTS was a significant source of TN in the catchment and identified rainfall as an influencing factor controlling TN influx into the river. However, the estimated nutrient influx to the Johnston River may not be conclusive as the study did not account for other sources of nutrient inputs in OWTS-served regions, such as animal waste and household-level fertilizer application.

Nutrients from OWTS can migrate with seepage, preferential flow paths, groundwater or surface runoff entering surface waters. This is particularly seen in coastal areas such as Florida, USA and Peel-Harvey catchment, Western Australia. The estimated TN loading to coastal waters in Florida from OWTS has been estimated to be as high as 5.5 kg/capita/year. The contribution of nutrients from OWTS to the eutrophication of the Peel-Harvey catchment has been established. Estimated TN and TP influx from OWTS to the Peel-Harvey catchment is 2.3×10^4 and 1.6×10^3 kg/year respectively. Despite overseas and Australian case studies showing significant amounts of nutrient loadings to the surface waters, studies are yet to be undertaken in the context of OWTS as a potential source of nutrients in Queensland waters. Further, the impact on environmental values related to OWTS as a source of nutrient contamination in Queensland waters is yet to be comprehensively investigated.

Similarly, OWTS is one of the major sources of microbial pathogens in the environment. Numerous case studies have identified the relationship between OWTS and disease outbreaks. This includes diseases caused by drinking water and recreational water contamination. Compared to outbreaks associated with drinking water contamination, outbreaks associated with recreational water contamination in Queensland are poorly understood. Human pathogens have been identified as a cause of coral diseases such as white-band disease in the Florida Keys and the Caribbean Islands. Although these diseases are reported in Australia, their link to OWTS effluent disposal is yet to be investigated.

Chapter 3. Factors influencing total nitrogen dispersal from OWTS

3.1 Introduction

On-site wastewater treatment systems (OWTS) rely on natural processes to remove total nitrogen (TN) in wastewater. These include physical and biochemical processes that start in the treatment chamber and continue in the wastewater disposal area. Physical processes that remove settleable and floatable materials occur in the treatment chamber. Settleable materials form sludge at the bottom of the treatment chamber, while floatable materials form scum at the top. Biochemical processes such as anaerobic digestion break down organic matter into basic organic compounds such as volatile fatty acids and methane. Treatment chamber wastewater rich in organic compounds, nutrients and pathogens is further treated in the wastewater disposal area, which can be either a soil absorption system or a surface irrigation system. Wastewater treatment in the disposal area may include physical processes such as volatilisation, adsorption, desorption, leaching and filtration, and biochemical processes such as assimilation, nitrification, denitrification, plant uptake, anammox and feammox. Feammox is a process where NH_4^+ oxidation occurs under iron reducing conditions, with iron oxides [ferric iron, Fe(III)] as the electron acceptor. In this reaction, Fe(III) is reduced to ferrous iron Fe(II), while NH_4^+ is transformed to NO_2^-, nitrogen gas (N_2), or other nitrogen forms.

Natural processes that OWTS rely on for wastewater treatment are heavily dependent on the treatment system characteristics, wastewater characteristics as well as site characteristics. For example, in an anaerobic treatment chamber (e.g. septic tank), methanogenesis and mineralisation of organic matter occur, whereas, in an aerobic treatment chamber, nitrification and organic matter oxidation occur. This chapter discusses factors influencing TN removal from wastewater in OWTS with particular emphasis on the soil absorption system. A summary of current state-of-the-art knowledge and knowledge gaps relating to TN removal by OWTS is provided in Table 3.1.

Table 3.1 Summary of what is known and unknown about parameters influencing nitrogen processes in soil absorption systems

	Parameter	Optimum conditions/relationships						
		Mineralisation	**Nitrification**	**Denitrification**	**Anammox**	**Feammox**	**Assimilation**	**Adsorption**
System Characteristics	Inflow Rates/ TN in inflow	SKG	SKG	(Negatively) (Trowsdale and Simcock, 2011)	N/A	SKG	N/A	(Negatively)
System Characteristics	Hydraulic residence time	(Positively) (USEPA, 2002).	(Positively) (Peterson et al., 2015)	(Positively) (Peterson et al., 2015)	N/A	N/A	N/A	N/A
Wastewater Characteristics	pH		7.5 – 8 (Hossain et al., 2010; Keen, 1984)	> 6.5 (Prakasam & Loehr, 1972; Thomas et al., 1994)	Acidic conditions?	Acidic conditions?	SKG	N/A
Wastewater Characteristics	DO (mg/L)	< 0	0.2 – 1.5 (Akaboci et al., 2018; Laureni et al., 2019; Yang et al., 2017)	< 0	0 – 1 (Robertson et al., 2012)	< 0	N/A	N/A
Wastewater Characteristics	Eh	< 0 SKG	> 0 SKG	> 0 SKG	< 0 SKG	< 0 SKG	N/A	N/A

Notes:

SKG – significant knowledge gap; ( ) – correlated; N/A – not applicable; and Eh – Redox potential

	Parameter	Optimum conditions/relationships						
		Mineralisation	Nitrification	Denitrification	Anammox	Feammox	Assimilation	Adsorption
Site Characteristics	Degree of Saturation (DS)	N/A	(Positively)	(Positively) (Willard et al., 2017)	(Positively) ?	(Positively)?	N/A	N/A
	Organic matter (OM)	N/A	(Positively)	SKG	(Positively) ?	SKG	(Positively)?	(Negatively) ?
	Temperature (℃)	28 (USEPA, 2002)	SKG	28 – 30 (Chen et al., 2012; Yang et al., 2011)	SKG	SKG	(Positively)?	N/A
	Soil Texture/ Type	N/A	High in fine grain soils?	High in fine grain soils (Liu et al., 2014; Waller et al., 2018)	High in fine grain soils	SKG	SKG	Increase with clay content?
	Alternating dry – wet conditions	N/A	SKG	Ambiguous	High in the wet season	SKG	High in the wet season?	Low in the wet season?
	TN removal Rate (mg/kg/day)	SKG	SKG	SKG	N-NH_4 = 0.5 TIN = 1.0 (Robertson et al., 2012)	1.2 ± 0.3×10^{-3} (in agricultural soils; Yang et al., 2012)	Depends on vegetation	SKG
Processes	TN transformation /removal	SKG	SKG	0 – 35% (Ritter and Eastburn, 1988)	80% of 100 mg/L effluent (Robertson et al., 2012)	SKG	Depends on vegetation	SKG
	Mediator	Ordinary heterotrophic organisms	*Nitrosomonas; Nitrosopira, Nitrosovibrio, Nitrosococcus Nitrobacter, Nitrospira, Nitrococcus, Nitrocystis, Nitrospina*	*Alcaligenes, Bacillus, Chromobacterium, Neisseria, Paracoccus, Pseudomonas*	*Candidatus* genera (Alba Pedrouso et al. (2019)	Fe^{3+} (Yang et al., 2012)	N/A	N/A

Notes:

SKG – significant knowledge gap; ( ) – correlated; and N/A - not applicable

3.2 Treatment Chamber Wastewater Treatment

3.2.1 Physical treatment processes

Physical treatment includes sludge formation at the bottom and scum formation at the top of the treatment chamber water column. The sludge is formed by sedimentation of settleable materials such as suspended solids and organic matter. Floatable materials such as grease and oil form the scum (Figure 3.1). The physical treatment removes about 60 – 80% of settleable and floatable matter in wastewater. This significantly impacts the wastewater treatment processes in OWTS by:

- ensuring a clear wastewater stream discharge into the wastewater disposal area;
- reducing the organic matter load to be removed in the wastewater disposal area;
- preventing clogging in the wastewater disposal area;
- enhancing the disinfection (if necessary) process.

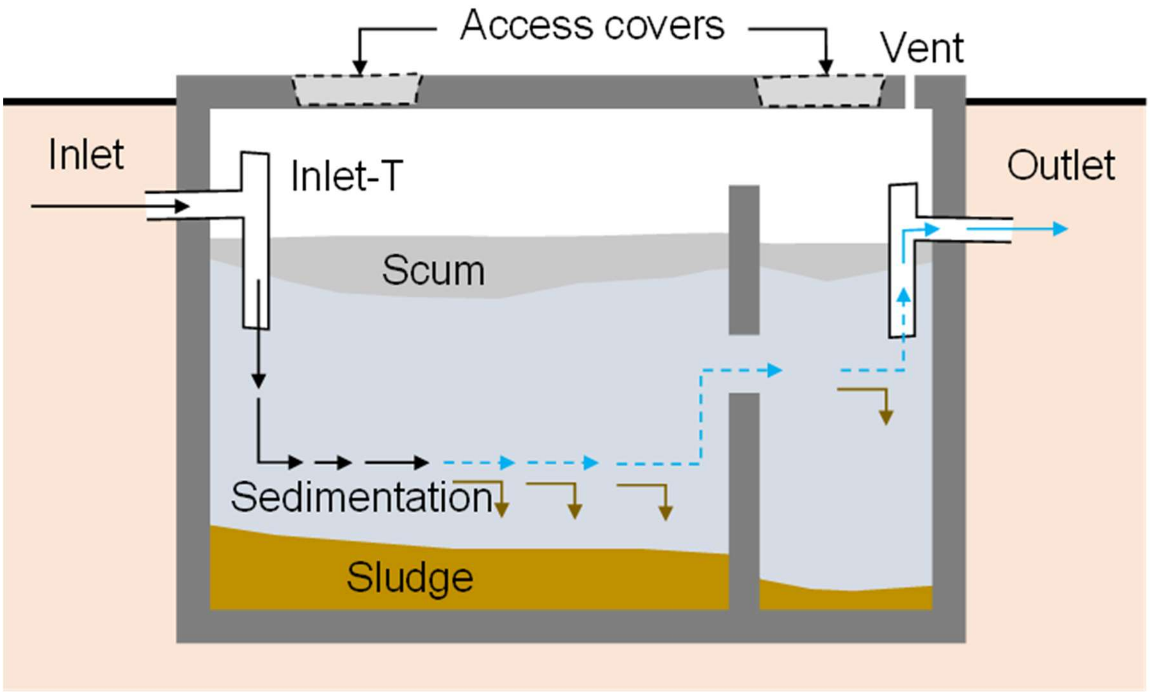


Figure 3.1 Sedimentation and scum formation in the treatment chamber

3.2.2 Biochemical treatment processes

a) Anaerobic treatment chamber processes

Organic matter in the treatment chamber includes proteins, carbohydrates, and lipids. In the treatment chamber, organic matter breaks down into smaller compounds such as ammonia and volatile fatty acids and microbial-driven processes that are governed by oxygen availability (Elmitwalli, 2013). Substantial variations in the concentrations of NH_4^+ and BOD present in an anaerobic and aerobic treatment chamber effluent are evident from the data summarised in Table 3.2 which demonstrates the influence of oxygen in primary treatment. Modified versions of a septic tank such as an up-flow anaerobic sludge blanket (UASB), high-rate UASB reactors and Imhoff tanks can achieve high BOD removal compared to conventional septic tanks due to different wastewater flow mechanisms taking place. However, these modified systems show similar TN removal efficiencies to a conventional septic system (Table 3.2). Energy demand and sophisticated technology that demands regular maintenance are among the other disadvantages of modified septic systems.

Table 3.2 Characteristics of effluent from different types of treatment chambers

Type	BOD (mg/L)	Total N (mg/L)	N_{Kj}-N* (mg/L)	NH_4-N (mg/L)	References
Conventional septic tank	300 – 600	40 – 90	-	-	Henze and Ledin (2001)
	94		44		USEPA (2002)
	-	-	-	106 ± 17	Robertson et al. (2012)
	-	50 – 84	-	42 – 76	De and Toor (2016)
	-	-	-	25.1	Garcia et al. (2013)
Aerobic treatment system	-	-	-	0.0911	Garcia et al. (2013)
	5-50	25 - 60	-	-	USEPA (2002)
USAB septic tank	115 – 400	20 – 80	-	-	Henze and Ledin (2001)
	197- 321	-	53 - 77	50 - 65	Al-Shayah and Mahmoud (2008)

*Total kjeldhal nitrogen

However, the complete conversion of BOD to methane (methanogenesis, Eq 3.1) does not occur in the treatment chamber. Maximum methanogenesis observed in septic tanks is 50% and 60% for domestic wastewater and blackwater, respectively (Elmitwalli, 2013). This is because conditions in the treatment chamber are not ideal for methane production (USEPA, 2002). The optimum temperature for methane-producing bacteria is around 28 ℃, which is above the average temperature of OWTS in most parts of the world including Queensland, Australia.

Methanogenesis can significantly reduce treatment chamber maintenance frequency by breaking down the settleable materials in wastewater (Elmitwalli, 2013). Baumann et al. (1978) found that the process can reduce sludge and scum volume by approximately 40%. However, high BOD removal may impact biological TN removal in the soil absorption system. This is because denitrifying bacteria are heterotrophic and require organic matter to meet their energy needs. A detailed discussion on heterotrophic denitrification and how it is influenced by organic matter content in wastewater is presented in Section 3.3.1.

$$CH_3COO^- + H^+ \rightarrow CH_4 + CO_2 \qquad \text{(Eq 3.1)}$$

Anaerobic digestion of proteins into NH_4^+ in the treatment chamber occurs in a two-step biological process, namely, hydrolysis (Eq 3.2) and fermentation (Eq 3.3, Rout et al., 2021). Raw wastewater generally consists of a high concentration of ammonia. With ammonification, the ammonia concentration in the treatment chamber effluent may increase up to 85% (USEPA, 2002). Although mineralisation itself is not a TN removal mechanism, it plays an important role in biological TN removal by converting organic nitrogen to NH_4^+.

$$Protein \rightarrow (R^+ - NH_2) + R^+ - OH \qquad \text{(Eq 3.2)}$$

$$(R^+ - NH_2) + H_2O \rightarrow NH_3 + R^+ - OH \qquad \text{(Eq 3.3)}$$

where R^+ represents the carbon monomer of the amino acid.

Anaerobic digestion in the treatment chamber includes several other intermediate anaerobic processes. It includes the hydrolysis of lipids, anaerobic oxidation and acidogenesis (Figure 3.2). All the processes illustrated in Figure 3.2 may also occur in the upper parts of the soil absorption system, just above the biomat where anaerobic conditions exist. However, processes other than methanogenesis and ammonification do not significantly impact the net TN removal process in OWTS and therefore, are not discussed in this report.

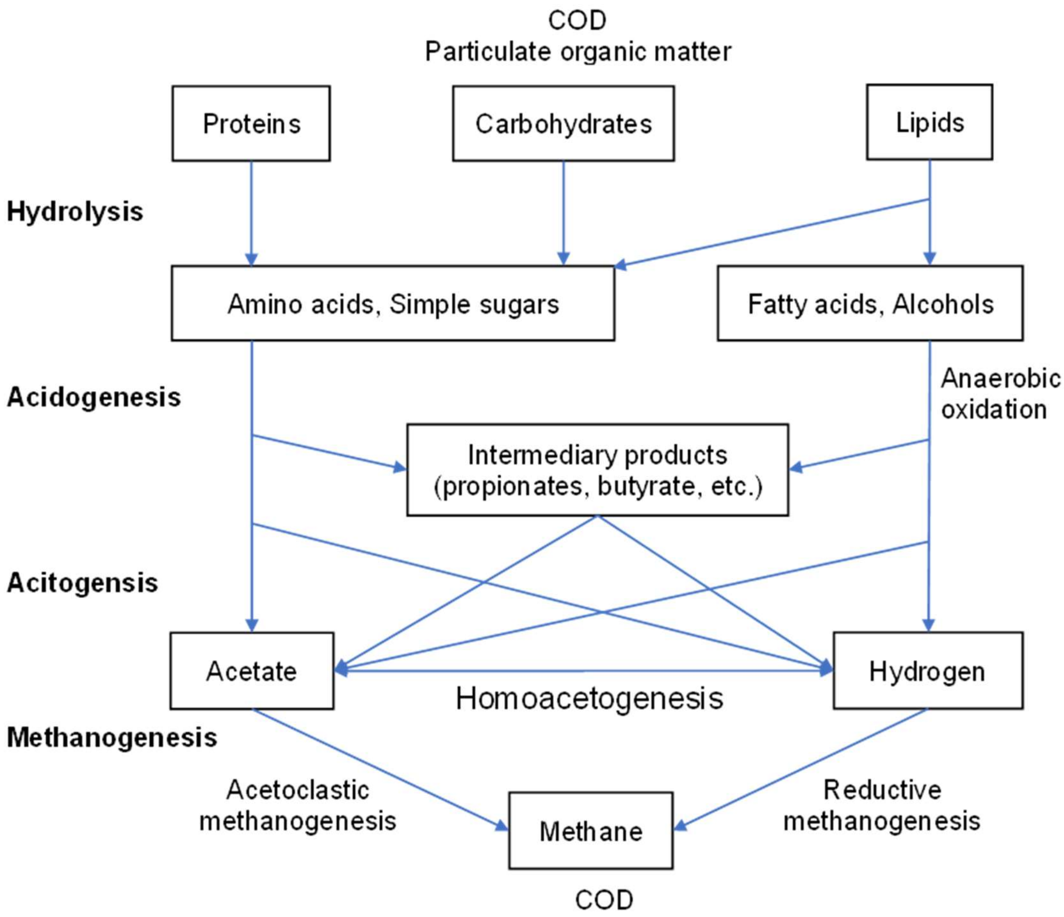


Figure 3.2 Anaerobic degradation phases of organic matter (Henze et al., 2008; Luostarinen & Rintala, 2005)

b) Aerobic treatment system processes

Aerobic treatment systems are pre-engineered to oxidise wastewater by aeration. The systems may include suspended growth elements, fixed growth elements or both (USEPA, 2002). Similarly, aerobic filters such as sand filters, biofilters and packed bed sand filters are also used in OWTS to achieve improved wastewater treatment. However, aerobic systems generally tend to produce more sludge.

In the presence of oxygen in aerobic treatment systems, complete hydrolysis of organic matter is expected. The processes may include organic matter oxidation (Eq 3.4), nitrification (Eq 3.5) and sulphide oxidation (Eq 3.6). This makes the nitrogen species composition in an aerobic treatment chamber effluent significantly different from an anaerobic treatment chamber effluent. For example, NH_4^+ concentration in the anaerobic treatment chamber effluent is in the range of 25 – 106 mg/L, whereas in the aerobic treatment chamber effluent, it is around 0.09 mg/L (Table 3.2) as a result of nitrification (Eq 3.5). Though the nitrogen species composition is different, effluent from both treatment chamber types will have similar TN concentrations (20 – 90 mg/L, Table 3.2). This highlights the fact that regardless of the design, limited TN removal occurs in the treatment chamber.

$$CH_2O + O_2 \rightarrow CO_2 + H_2O \quad \text{(Eq 3.4)}$$

$$2\,NH_4^+ + 4\,O_2 \rightarrow 2\,NO_3^- + 2\,H_2O + 4\,H^+ \quad \text{(Eq 3.5)}$$

$$H_2S\ (or\ organic\ sulphates) + 2O_2 \rightarrow SO_4^{2-} + 2\,H^+ \quad \text{(Eq 3.6)}$$

Regardless of the characteristics of the treatment train, OWTS eventually disperse treatment chamber effluent (partially treated wastewater) into the environment. Once disposed into the environment, several physicochemical and biochemical processes occur in relation to the TN in the effluent. The discussion below presents the key TN removal processes and factors influencing these processes in soil absorption systems.

3.3 Wastewater TN Removal Processes in the Soil Absorption System

Soil absorption systems are generally used to disperse partially treated wastewater into the environment (Beal et al., 2005). As wastewater infiltrates and percolates through the unsaturated soil layer (vadose zone), various processes transform and recycle wastewater constituents before reaching the saturated zone and subsequently the groundwater. These include physical, physicochemical, and biochemical processes. Physical processes, such as filtration and straining, play a key role in removing organic matter, total suspended solids and pathogens. Ion exchange, precipitation and adsorption are some of the physicochemical processes involved in immobilising PO_4^{3}, NH_4^+, Ca^{2+} and Na^+ in wastewater. Biochemical processes include nitrification, denitrification, microbial assimilation and plant uptake. Both, physicochemical and biochemical TN removal processes are impacted by several factors, including construction and design practices as well as wastewater and site characteristics. Key wastewater TN removal processes in the soil absorption system are illustrated in Figure 3.3 and subsequent sections discuss the influence of the above noted factors on these processes with particular attention to Queensland conditions.

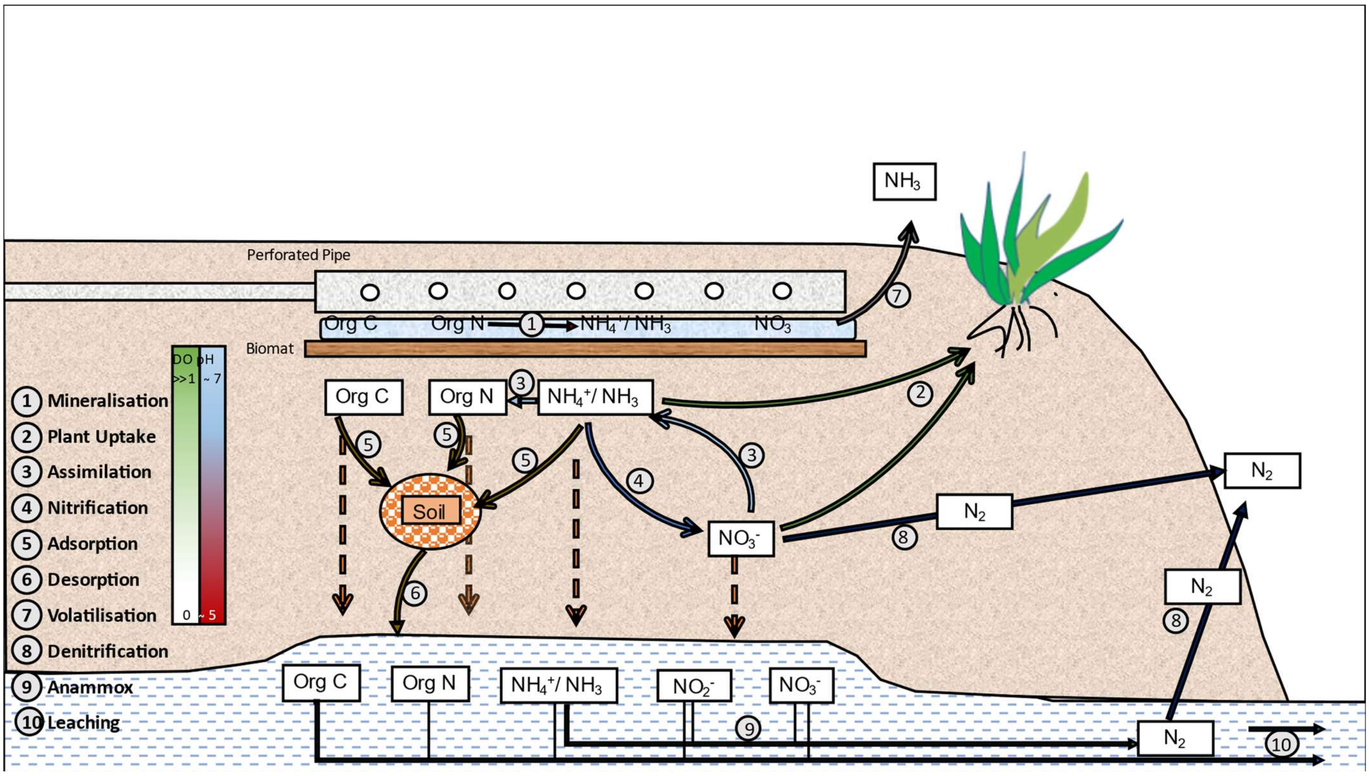


Figure 3.3 Nitrogen removal processes in a soil absorption system at the subsurface

3.3.1 TN removal at the biomat

A biomat is formed on the infiltrative surface due to the accumulation of microorganisms such as bacteria and protozoa, biodegradable organic matter and total suspended solids. Also known as a clogging mat, the biomat filters suspended solids and microorganisms and controls the hydraulic infiltration rate. Figure 3.4 schematically illustrates the wastewater treatment processes in a soil absorption zone, including at the biomat. Straining and filtration in a mature biomat can remove 70 – 95% of organic matter (Daniel & Bouma, 1974; Pell & Nyberg, 1989; Van Cuyk et al., 2001). The biomat also removes relatively large pathogens, such as helminths and bacteria.

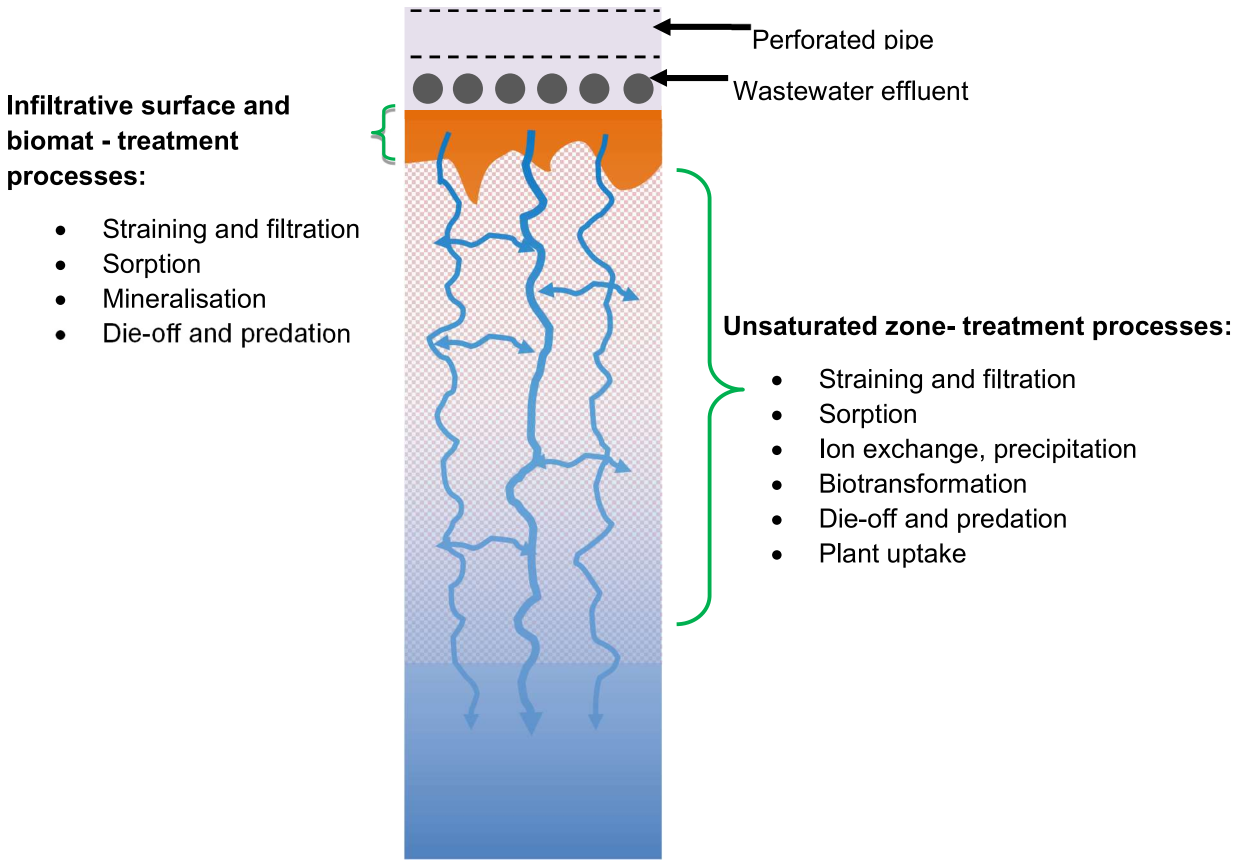


Figure 3.4 Schematic of soil absorption system treatment processes (Adapted from Siegrist et al., 2000)

Although the direct contribution of the biomat to nutrient removal is not fully understood, its indirect role in TN removal is well known. The resistance of a mature biomat can reduce hydraulic conductivity creating saturated conditions above and an unsaturated zone below it (Beal et al., 2006; Siegrist & Van Cuyk, 2001). In the saturated zone above the biomat, anaerobic digestion of organic nitrogen and conversion to NH_3/NH_4^+ occurs. Unsaturated conditions below the biomat are essential to convert wastewater NH_3/NH_4^+ to NO_3^- (nitrification), which can then be denitrified in the deeper layers of the subsoil.

The characteristics of the biomat, such as age, thickness, saturated hydraulic conductivity and resistance can influence its performance. It takes about a year from the start of operation for a mature biomat to establish that can result in a steady long-term vertical flow rate (Beal et al., 2005). Therefore, a biomat younger than a year may not be able to support TN removal effectively. However, with continuous operation for several years, a considerable amount of organic matter may accumulate resulting in a thick, highly resistant biomat. In most cases, the low hydraulic conductivity of a thick, high resistance biomat is associated with effluent ponding on the ground surface (Baveye et al., 1998; Beach et al., 2005; Bouma, 1975). Referred to as hydraulic failure, this may result in a nitrogen surcharge into the surroundings.

3.3.2 Nitrification

Nitrification is the process during which nitrifying bacteria fulfil their energy requirements (autotrophic) by oxidising NH_4^+ into NO_3^-. The process reduces the oxygen demand of wastewater as well as reduces the ammonia toxicity to the receiving environment (Okabe et al., 2011). Nitrification occurs in two phases: (1) oxidising NH_4^+ to nitrite (NO_2^-, Eq 3.7) by ammonia-oxidising bacteria (AOB); and (2) oxidising NO_2^- to NO_3^- by nitrite-oxidising bacteria (NOB, Eq 3.8). Both AOB and NOB have slow growth rates and are sensitive to ambient environmental conditions such as temperature, pH, DO concentration, alkalinity, chemical oxygen demand/total Kjeldahl nitrogen ratio and the presence of any toxic chemicals present (Okabe et al., 2011).

$2\ NH_4^+ + 3\ O_2 \rightarrow 2\ NO_2^- + 4\ H^+ + 2\ H_2O$ (Eq 3.7)

$2\ NO_2^- + O_2 \rightarrow 2\ NO_3^-$ (Eq 3.8)

A few studies have reported high nitrification rates within 50 cm below the surface (Cogger, 1988; Kristiansen, 1981; Pell & Nyberg, 1989; Walker et al., 1973). This zone is generally unsaturated and moist due to the resistance and limited hydraulic conductivity of the biomat, making conditions favourable for aerobic nitrifying bacteria to grow. Therefore, high nitrification rates in the upper part of the soil absorption system are possible. Li et al. (2011) investigated the presence of nitrifying bacteria at 25 different sampling points in a soil absorption system. The distribution of the sampling points in this study is given in Figure 3.5, and the density of nitrifying bacteria is given in Table 3.3. The high nitrifying bacteria density close to the surface may explain the high nitrification rates observed within the first 50 cm by past researchers, as noted above.

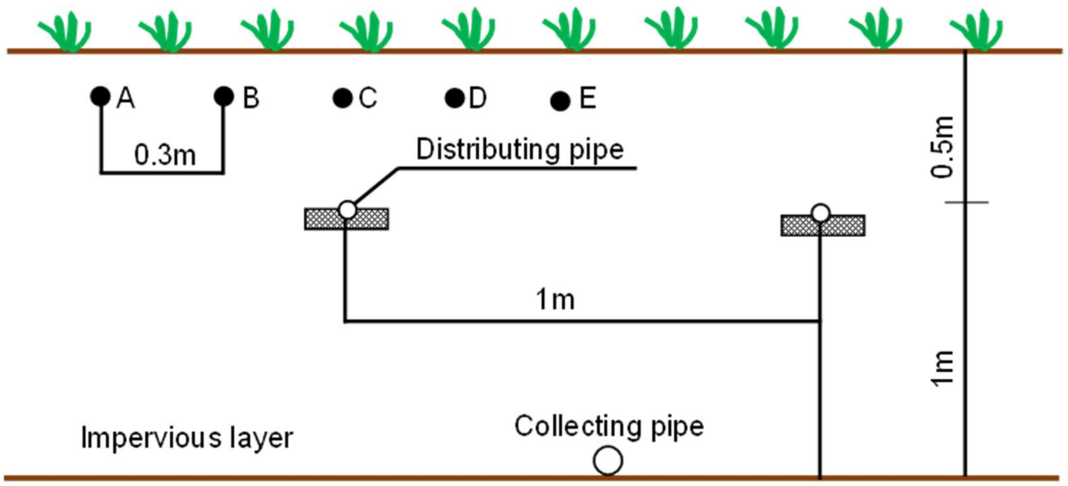


Figure 3.5 Distribution of sampling points in the soil absorption system in the study by Li et al. (2011)

Table 3.3 Distribution of nitrifiers in a soil absorption system (Source: Li et al., 2011)

Sample point	**Nitrifying bacteria population x10^5**				
	0.3 m	0.5 m	0.7 m	0.9 m	1.1 m
A	3.5 ± 1.2	1.6 ± 1.4	0.58 ± 0.04	0.033± 0.01	0.006 ± 0.001
B	55 ± 1	42 ± 2	7.6 ± 1.5	4.5 ± 0.4	0.78 ± 0.04
C	740 ± 120	670 ± 500	63 ± 1	53 ± 2	8.9 ± 0.2
D	43 ± 1	35 ± 2	4.6 ± 0.5	3.4 ± 0.2	0.64 ± 0.02
E	6.6 ± 1.4	2.6 ± 1.8	0.36 ± 0.07	0.2 ± 0.02	0.05 ± 0.002

In addition to autotrophs, a few heterotrophs (organisms that use organic carbon as an energy source) may also contribute to nitrification. Heterotrophic nitrification can produce nitrous oxides (NO and N_2O) and nitrogen gas (Eq 3.9 and 3.10, Skiba and Smith, 1993). However, this is a slow process compared to autotrophic nitrification. Therefore, even under appropriate conditions, complete TN removal by heterotrophic nitrification cannot be expected.

$2.5\ CH_3COOH + 4\ NO_3^- + 4\ H^+ \rightarrow 2\ N_2 + 5\ CO_2 + 7\ H_2O$ (Eq 3.9)

$1.5\ CH_3COOH + 4\ NO_2^- + 4\ H^+ \rightarrow 2\ N_2 + 3\ CO_2 + 5\ H_2O$ (Eq 3.10)

Heterotrophic nitrification occurs under harsh environmental conditions. Skiba and Smith (1993) found that the process occurs particularly under dry conditions. Acidic soil conditions are also required for heterotrophic nitrification (de Boer & Kowalchuk, 2001; Duggin et al., 1991). In Queensland, about two third of soils are acidic (Ahern et al., 1994). Therefore, anoxic heterotrophic nitrification may play a key role in soil absorption systems in Queensland. However, the process in soil absorption systems in Queensland or other parts of the world is poorly understood. To the best available knowledge, the influential factors, rates, nitrogen gas production and related TN loss due to heterotrophic nitrification are yet to be comprehensively investigated. This knowledge gap is a constraint for conducting an accurate nitrogen mass balance for OWTS.

3.3.3 Denitrification

Denitrification is another key TN removal process in a soil absorption system. The process is an essential step in biological TN removal, which converts nitrate produced during nitrification to nitrogen gases (N_2 and N_2O; Eq 3.11). This multi-step process is driven by anoxic, heterotrophic bacteria. Anaerobic conditions, hence, denitrification generally occurs deep in the soil absorption system. This was confirmed by the high denitrifying bacteria density observed deep in the soil absorption system in the study undertaken by Li et al. (2011). The high denitrifying bacteria density between 0.7 – 1.1 m in the soil absorption system (Table 3.4) indicates that denitrification is the dominant process at these depths (Li et al., 2011). However, these observations may not apply to all soil absorption systems as bacteria distribution is significantly impacted by the characteristics of the soil absorption system matrix. In their experimental study, Li et al. (2011) used a matrix comprising 65% local brown soil, 30% coal sludge and 5% dewatered sludge. Soil porosity, moisture and organic matter content may impact the bacteria distribution and thereby the nitrification-denitrification process. Detailed discussion on factors influencing the nitrification-denitrification process is provided in Section 3.4.

$$4\ NO_3^- + 5\ CH_2O + 4\ H^+ \rightarrow 2\ N_2(g) + 5\ CO_2 + 7\ H_2O \qquad \text{(Eq 3.11)}$$

Table 3.4 Distribution of denitrifying bacteria in a soil absorption system based on the study by Li et al. (2011)

Sample point	Denitrifying bacteria population ×10^6				
	0.3 m	0.5 m	0.7 m	0.9 m	1.1 m
A	6.6 ± 2.1	9.8 ± 1.9	15 ± 2	60 ± 3	0.001± 0.003
B	28 ± 15	52 ± 8	330 ± 60	770 ± 40	3500 ± 400
C	680 ± 50	960 ± 120	4500 ± 100	8800 ± 100	76000 ± 2000
D	58 ± 7	69 ± 2	180 ± 30	780 ± 30	4800 ± 300
E	4.1 ± 1.2	7.3 ± 1.5	39 ± 4	59 ± 14	290 ± 140

3.3.4 Assimilation

Plants and microorganisms may uptake nitrate and ammonia in a soil absorption system (Huang et al., 2000). The nitrogen mineralised by microorganisms is added back to the subsurface with die-off. Therefore, assimilation by microorganisms cannot be considered a permanent or significant TN removal mechanism. On the other hand, plants can take up and fix nitrogen. OWTS generally disperse nitrates more than 1 m below the surface. Therefore, deep-rooted plants can be used for TN removal in a soil absorption system. However, plant uptake as a potential TN removal mechanism in a soil absorption system is yet to be investigated in-depth.

3.3.5 Ammonification

Ammonia volatilisation is another mechanism that may remove TN from a soil absorption system. However, the process requires specific conditions. Hence, TN loss due to volatilisation may be insignificant. Optimum ammonification occurs at temperatures above 28 ℃ and pH greater than 9.3 (USEPA, 1998). The limited knowledge of ammonification as a potential TN removal mechanism in a soil absorption system is attributed to the need for specific conditions.

3.3.6 Anammox

The occurrence of anaerobic ammonium oxidation (anammox) has been identified in wastewater treatment systems by several researchers (Siegrist et al., 1998; Verstraete & Philips, 1998). This anoxic heterotrophic microbial process occurs in the presence of organic carbon, ammonia, and nitrate (Eq 3.12). The process is dominant in low temperatures. Robertson et al. (2012) observed anammox in groundwater, below a soil absorption system. The study reported 80% TN removal due to anammox in the Long Pine campground septic system on the north shore of Lake Erie, Canada (Robertson et al., 2012). Although groundwater is not part of the soil absorption system, understanding the process is essential, particularly in the context of risk assessment in relation to OWTS. However, anammox needs anoxic conditions. Currently, there is no strong evidence to suggest that it is a major process involved in TN removal, but merits future research attention.

$$3\ NO_3^- + 5\ NH_4^+ \rightarrow 4\ N_2 + 9\ H_2O + 2\ H^+ \qquad \text{(Eq 3.12)}$$

3.3.7 Adsorption

Ammonium adsorption can be a significant TN immobilisation process, particularly in agricultural lands. Here, positively charged ammonium is adsorbed to negatively charged soil (clay) particles. Although studies are limited, it is understood that ammonium adsorption is not a significant process in the context of TN removal from OWTS. This is because of the wastewater composition. Wastewater is rich in Ca^{2+}, Mg^{2+} and Na^+ ions which compete with ammonium for adsorption to the soil profile (Kannan & Parameswaran, 2021). Therefore, it can be hypothesised that TN removal due to ammonium adsorption may be limited in soil absorption systems.

3.3.8 Chemical (abiotic) TN removal processes

Other than the processes noted above, minor processes could also contribute to TN removal in the subsurface. This includes abiotic nitrate reduction (chemo-denitrification, Wang et al., 2020). This process may occur under anoxic conditions in the presence of electron donors such as Mn^{2+} and Fe^{2+} (Luther et al., 1997; Reddy et al., 2013). Chemo-denitrification in the presence of Mn^{2+} as an electron donor is illustrated in Eq 3.13 – 3.15. The process is heavily dependent on Fe^{2+}, nitrite, nitrate and organic carbon concentration in the soil (Wang et al., 2020) and found to occur in agricultural soils in intensively irrigated areas in northern Queensland (Reading et al., 2019). However, the importance of chemo-denitrification in TN removal from OWTS in Queensland or elsewhere is yet to be comprehensively understood.

$$5\ Mn^{2+} + 2\ NO_3^- + 4\ H_2O \rightarrow 5\ MnO_2 + N_2 + 8\ H^+ \qquad \text{(Eq 3.13)}$$

$$5\ Mn^{2+} + 2\ NO_3^- + 8\ HO^- \rightarrow 5\ MnO_2 + N_2 + 4\ H_2O \qquad \text{(Eq 3.14)}$$

$$3\ MnO_2 + 2\ NH_3^+ + 6\ H^+ \rightarrow 3\ MnO_2 + N_2 + 8\ H^+ \qquad \text{(Eq 3.15)}$$

3.4 Factors Influencing TN Removal in Soil Absorption Systems

3.4.1 System characteristics

a) Design parameters

The wastewater treatment processes in the treatment chamber are mainly influenced by its design parameters. This includes dimensions, inlet and outlet design, residence time and compartmentalisation. The influence of these design parameters on the physical wastewater treatment process is well understood and has been used to develop recommendations for OWTS designs. For example, 6-24 hrs of hydraulic residence time is required for optimum sedimentation (Baumann & Babbit, 1953). Therefore, septic tanks are recommended to have sufficient capacity to provide a minimum 24 hrs hydraulic retention (USEPA, 2002). Table 3.5 provides a brief overview of design considerations for an OWTS treatment chamber.

However, the influence of OWTS design guidelines on TN removal is not clear due to several reasons. Firstly, guidelines are only developed considering the organic matter and pathogen removal and may not reflect the conditions required for TN removal. For example, the setback distances act as a buffer zone between the wastewater treatment system and sensitive environments such as groundwater and nearby surface waters. Appropriate guidelines for setback distances are provided considering the retention and die-off of both, harmless gut organisms and pathogens (AS/NZS 1547:2000, 2000). Preventing nutrient migration to groundwater in sensitive environments is not considered in specifying setback distances. Secondly, treatment chamber parameters are designed mainly to achieve optimum organic matter removal. Since nitrifying and denitrifying bacteria use organic matter as their energy source, organic matter removal may have an impact on the TN removal processes in a soil absorption system. The influence of organic matter on TN removal is provided in Section 3.4.2.

Table 3.5 Design considerations for septic tanks

Design Parameter	Recommendation	References
Volume	A volume sufficient to provide 24 hr hydraulic residence time.	Baumann and Babbit (1953)
Geometry	Length:Width ratio ≥ 3:1 to reduce short-circuiting of raw wastewater.	Ludwig (1950)
Compartmentalisation	A series of compartments provide better suspended solids removal compared to a single compartment.	Baumann and Babbit (1953)
Inlet and Outlet	Baffles to prevent solids from leaving the tank - 2-3 inch (50–75 mm) drop between inlet and outlet. - Minimum 1 inch (25 mm) clear space between the outlets. - Could use removable and cleanable effluent screens.	USEPA (2002)
Watertightness	Tanks must be watertight.	
Location	Must be located in accordance with setback distances. e.g: For OWTS less than 10 EP (AS/NZS 1547:2000, 2000) **Depth to groundwater** 0.6 m **Property boundary** 1.5 m **Bore wells** 15 m **Recreation area** 3 m **Surface water** 15 m	USEPA (2002)
Filters	Depending on the site and wastewater characteristics.	USEPA (2002)

b) Construction practices

The natural hydraulic conductivity of the soil can be impacted during construction. Disturbance to the natural soil may include compaction, smearing and deposition during construction and may reduce the hydraulic conductivity. Listed in Table 3.6 are construction-related issues that can impact hydraulic conductivity (Siegrist et al., 2000). The reduced hydraulic conductivity can result in hydraulic failure or clogging, which may lead to ponding and the surface surcharge of wastewater.

Table 3.6 Construction-related impacts that can influence wastewater treatment (Source: Siegrist et al., 2000)

Effect	Cause
Compaction	Vehicular traffic on the exposed infiltration surface
Smearing or puddling of the surface	Vehicular tyre or track at the infiltration surface interface
Deposition of wind-blown fines	While the infiltrative surface is exposed

c) Installation age and operational service life

OWTS are installed according to the knowledge available at the time. Therefore, older systems may be even less effective in removing TN compared to those installed more recently. Similarly, the hydraulic conductivity of the soil absorption system may reduce with long-term operation due to biomat formation. Further, soil clogging may result from treatment chamber effluent disposal over the long term (Dawes & Goonetilleke, 2003). Wastewater rich in sodium (Na^+) can increase the sodicity or the exchangeable sodium percentage (ESP) in the soil, distorting the soil structure which consequently will result in reduced pore size and drainable pore volume. Reduced pore spaces may reduce hydraulic conductivity leading to wastewater ponding on the surface, leading to hydraulic failure.

d) Hydraulic loading rate and hydraulic conductivity

In general, hydraulic loading rates (HLR) are designed to be a smaller fraction (1–10%) of the saturated hydraulic conductivity (K_{sat}) of the soil (Siegrist et al., 2004). The reason for having a fraction of K_{sat} as the design loading rate is to ensure unsaturated conditions in the subsurface, which is essential for effective wastewater treatment. Hydraulic infiltration rates in a soil absorption system changes with the operational age which can be broadly divided into three stages (Siegrist, et al., 2002). The first stage is characterised by nonuniform infiltration due to imperfect distribution networks. After a few months to a year of operation, the infiltration rate reduces to a lower fraction of K_{sat} due to soil clogging and the development of a biomat. Identified as operational Stage 2, this will continue for several years, during which capacity-limited processes such as P adsorption and NH_4^+ adsorption may become exhausted (Siegrist et al., 2004). Once the soil absorption system reaches operational Stage 3, infiltration rates will further reduce and reach the long-term acceptance rate for another 10-20 years.

Li et al. (2011) investigated the role of the hydraulic loading rate on TN removal using soil columns containing brown soil (65 %), coal slag (30 %) and dewatered sludge (5 %). The study found a decrease in NH_3 and TN removal with increasing hydraulic loading rates and recommended 81 mm/day for optimum TN removal (Li et al., 2011). According to the finding of this study given in Table 3.7, the highest TN removal of 94.5 % was observed at a 40 mm/day hydraulic loading rate. The TN and NH_3 removal decreased by 25.4 % and 17.6 %, respectively, when hydraulic loading rates increased from 40 mm/day to 100 mm/day. The researchers attributed this behaviour to reduced nitrification and denitrification resulting from increased hydraulic loading rates. Reduced nitrification can also affect the denitrification rates by limiting NO_3^- availability. However, the findings of Li et al. (2011) may not necessarily represent the TN removal rate for OWTS with a mature biomat. It is highly unlikely that a mature biomat could have formed during the four-month duration of this study.

Table 3.7 Effect of hydraulic loading rates on TN removal based on the study by Li et al. (2011)

Hydraulic loading rate (mm/day)	$N\text{-}NH_3$ removal (%)	TN removal (%)	COD (%)
40	97.4 ± 1.8	89.8 ± 1.8	-
65	92.8 ± 0.5	82.9 ± 2.9	-
81	87.7 ± 1.4	70.1 ± 4.0	84.8 ± 2.1
100	79.2 ± 2.9	60.5 ± 3.4	-

Beal et al. (2006) suggested that the resistance provided by a mature biomat governs the hydraulic infiltration rate in the soil absorption system over the unsaturated soil hydraulic conductivity. Siegrist et al. (2004) also noted that a mature biomat dominates the wastewater infiltration rate over the wastewater loading rate, particularly in systems located in low permeable natural soils such as dense clay or silty clay. In a 16-month-long study conducted using sand, ferrosol and vertosol soil columns and with varying loading rates, Beal et al. (2006) observed a four-order magnitude reduction in the saturated hydraulic conductivity and one-order magnitude reduction in the long-term vertical flow rate due to the presence of a mature biomat.

e) Operational practices

Operational practices can directly influence TN removal in a soil absorption system. Using a column study, De Vries (1972) demonstrated that resting the soil absorption system for some time would improve wastewater treatment. The researcher suggested resting for 7 to 9 days would allow the soil absorption system clogged with organic matter to recover and re-establish a desirable hydraulic loading rate. The consumption of oxygen and the production of CO_2 during the recovery period indicates the consumption of accumulated organic matter. Though, the exact impact of short-term resting of a soil absorption system on TN removal is not clear.

Long-term hold in operation may reduce TN removal in a soil absorption system. The biomat can deactivate during a long operational hold. This may occur in OWTS used in seasonal facilities such as holiday homes. After re-commencing operations, it will take some time for the biomat to re-establish. Until the biomat is fully established, its services may not be optimal, such that organic matter and ammonia may contaminate the groundwater (Robertson et al., 2012). On the other hand, the coexistence of organic matter, ammonia, and nitrate in wastewater plumes may remove TN by the anammox process (Robertson et al., 2012). Therefore, current knowledge is inadequate to assess the true impact of long-term resting of a soil absorption system on TN removal efficacy and related environmental impacts.

3.4.2 Wastewater characteristics

a) Dissolved oxygen concentration

Oxygen availability has a significant impact on TN removal in the subsurface. In an aerobic treatment chamber dissolved oxygen (DO) concentrations greater than 2 mg/L are recommended for complete nitrification to occur (How et al., 2018). The NH_4^+ concentration in the effluent from an anaerobic treatment chamber (septic tank) is in the range of 42–106 mg/L. To completely nitrify this load, about 265 - 652 mg/L of dissolved oxygen is required (Eq 3.7-3.8). However, the DO in septic tank effluent is around 0.2–0.3 mg/L (Stuth & Lee, 2001) and insufficient to oxidise wastewater NH_4^+ completely. Therefore, it is essential to have an aerated zone in the subsurface for complete nitrification to occur to prevent wastewater NH_4^+ from reaching the receiving aquatic environment.

However, the presence of an unsaturated zone below the biomat does not necessarily indicate the presence of an aerobic zone (Siegrist et al., 1983). Continuous operation and long operational age may significantly reduce oxygen availability in the unsaturated zone. In addition, the BOD in wastewater may consume oxygen in the subsurface. Although the relationship between oxygen availability (both subsurface and DO) and TN removal in the soil absorption system (anaerobic OWTS) is yet to be established, it is clear that achieving complete nitrification is a challenge.

b) Organic matter content

Organic matter content reduces as the wastewater infiltrates through the soil absorption system. It is reported that over 80% of the organic matter is removed as wastewater percolates down to 25 cm (Barford et al., 2017; Li et al., 2018). Biodegradable organic matter content can act as a major limiting factor for TN removal in a soil absorption system (Yang et al., 2022). Organic matter is essential for the heterotrophic denitrification process (Eq 3.11), and its limited availability may significantly affect TN removal (Ritter & Eastburn, 1988; Wang et al., 2021).

Several approaches have been successfully tested to supplement the organic matter content and thereby achieve high TN removal compared to a conventional soil absorption system (Bowman & Focht, 1974; Cochet et al., 1990; Ritter & Eastburn, 1988). One approach was to introduce organic matter to the deeper layers of a soil absorption system. Known as shunt distribution, the method pumped raw wastewater into the deeper layers of a soil absorption system to supplement the carbon demand of the denitrifying bacteria, which resulted in high TN removal in several studies listed in Table 3.8 (Li et al., 2018; Pan et al., 2013; Wang et al., 2010; Yang et al., 2022). However, introducing raw wastewater into the deeper layers of the soil absorption system may increase the risk of organic nitrogen and NH_3/NH_4^+ contamination as nitrification does not occur in those layers. Amongst the other disadvantages of shunt distribution are the complex installation and associated financial costs. Further, the feasibility of shunt distribution in sandy soils such as in coastal regions has not yet been established and this merits future research.

Table 3.8 Influence of organic matter content on TN removal

Study	Soil matrix used	Shunt Distribution	Increase in removal (%)	
			NH_4^+	TN
Wang et al. (2010)	Brown soil – 80 %, cinder – 20% (by weight)	1:1 at 0.7 m depth	5.7	10
Pan et al. (2013)	Brown soil – 80%, coal slag - 20 % (by weight)	1:1 at 0.7 m depth		13.3
Wu et al. (2019)	Gravel - 10 cm Mixed culture substrate – 80 cm (Soil – 80%, carbonized rice hull – 15%, zeolite – 5%) Soil – 5 cm	1:2 at 0.7 m depth		23.1
Li et al. (2018)	Farmland soil - 30 cm bio-substrate – 80 cm, coarse sand - 10 cm, gravel - 10 cm	C:N = 3		98.3

c) Sodium concentration

Na^+ ions (measured as sodium adsorption ratio SAR) in wastewater influence the structural stability of the soil in a soil absorption system, and consequently, the treatment process and TN removal. Na^+ in wastewater released to the soil solution can replace other exchangeable cations in soil particles such as Ca^{2+} and Mg^{2+}. This may result in swelling (dispersion) of soil particles and reduced pore spaces. Wastewater with high SAR (above 5) can adversely impact the soil structure in many soil types. Ion exchange can reduce hydraulic conductivity due to soil particle dispersion. Dawes (2005) observed a significant increase in soil Na^+ concentration and a decrease in soil Ca^{2+} over the years of operation of an OWTS which could result in the risk of swelling of clay particles and dispersion. Swelling and dispersion can reduce hydraulic conductivity leading to hydraulic failure. However, detailed knowledge of how hydraulic conductivity changes as a function of SAR is lacking. An in-depth understanding will help to identify soils suitable for use as soil absorption systems.

3.4.3 Site characteristics

a) Climate

The climate as an influential factor for TN removal in OWTS includes rainfall and soil temperature. Rainfall determines subsurface saturated conditions and the depth to the groundwater table that directly impacts TN removal by OWTS. Table 3.9 summarises past studies that have investigated the impact of rainfall and temperature on TN removal from OWTS. O'Driscoll et al. (2014) investigated the impact of rainfall on TN dispersal from septic systems in the Pamlico River Estuary in coastal Beaufort County, North Carolina, USA. The study found that rainfall not only impacts the concentration, but also the nitrogen species leached into groundwater as illustrated in Figure 3.6. This phenomenon can be explained by the saturation of the subsurface and the rise of the groundwater table with the infiltration of rainwater. As the soil pores fill with infiltrating rainwater, the subsurface becomes anaerobic making conditions unsuitable for nitrification. Limited nitrification results in a high amount of organic N and NH_4^+ leaching into the groundwater during the wet season when precipitation (P) exceeds evapotranspiration (ET) (P > ET) compared to the dry season (P < ET). In addition, the leaching of soil-adsorbed organic N and NH_4^+ can also contribute to their high amount in the wet season. Nevertheless, low TN concentration in groundwater observed under P > ET conditions may not necessarily reflect increased denitrification rates, but rather dilution in increased groundwater volume. This highlights the need for considering TN loading to the groundwater instead of concentration when investigating TN removal by OWTS. This is particularly important in regions where high rainfall and groundwater movement is common.

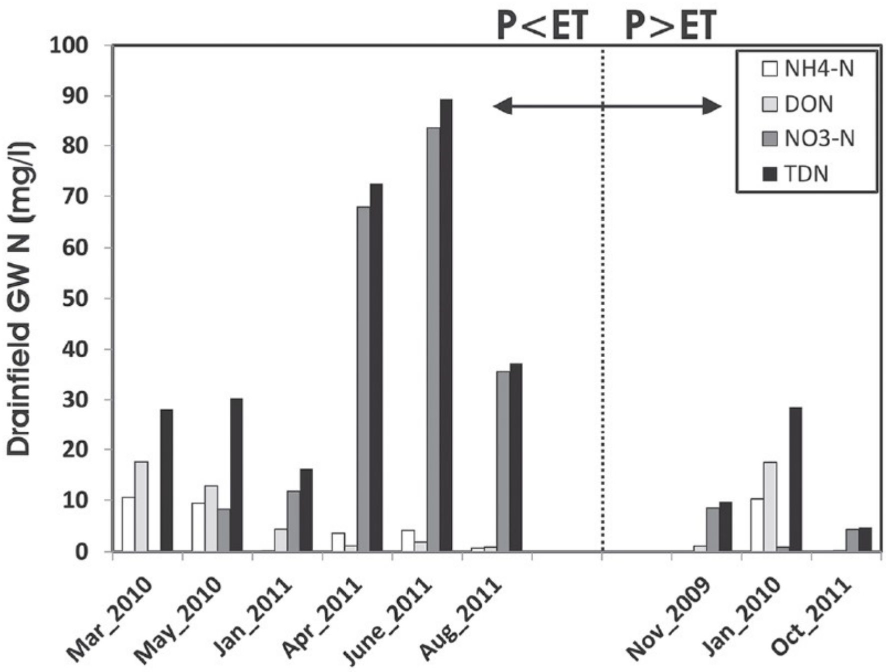


Figure 3.6 Variation in the concentration of different nitrogen species in the groundwater from the study by O'Driscoll et al. (2014)

Sexstone et al. (1985) investigated the variation of denitrification rates in sandy loam and clay loam soils after episodes of irrigation and rainfall using both, laboratory and field investigations. Followed by wetting events, increased denitrification rates were observed in sandy loam soils which were attributed to the establishment of an anaerobic zone where denitrification can occur. However, a prolonged wet season with saturated soil conditions can reduce the denitrification rate by limiting nitrification.

Table 3.9 Impacts of climatic variations on TN removal in OWTS

Study	Findings (TN removal)	Comments
Arnade (1999) Impact of rainfall on NO_3^- transport to groundwater	Considerable decrease in NO_3^- during the wet season.	Could be the result of dilution.
Cogger and Carlile (1984) Influence of seasonal variation on OWTS treatment performance	High TN transport in saturated soils	Low TN removal due to limited availability of NO_3^- due to low nitrification.
O'Driscoll et al. (2014) Seasonal variation in the quality of groundwater impacted by septic systems	Dry season (~ 1 m to groundwater): • TN >10 mg/L • NO_3^- dominated TN • Wet season: • TN < 10 mg/L • NH_3 and Org N dominated TN	Changes in oxic-anoxic conditions during the dry and wet seasons may explain the variation in N species. Low TN concentration in the wet season could be due to dilution.
Yuan et al. (2013) The effect of temperature on wastewater treatment in a soil absorption system using soil columns	NH_3-N removal > 97.1% at temperatures >13℃ TN removal >75% at temperatures >25℃, 85% at 33℃	In Queensland, these impacts may be negligible due to relatively high soil temperature.

Soil temperature is another significant factor that controls TN removal. This is because of the temperature sensitivity of nitrifying bacteria. Nitrosomonas, the bacteria that convert ammonia to nitrite, function in the temperature range between 5 – 45 ℃. However, nitrification rates decrease in temperatures above 30 - 35 ℃. Low growth rates of nitrobacter which convert nitrite to nitrate at a temperature over 30 ℃ can explain this behaviour. Nitrobacter is only functional between 5 – 30 ℃ (USEPA, 2002). Yuan et al. (2013) demonstrated the impact of temperature on the removal of NH_4^+ and TN in wastewater using a year-long column study and concluded that temperature above 13 ℃ ensures high nitrification and denitrification. In a laboratory experiment conducted using six different types of sand, Maag and Vinther (1996) found maximum nitrification rates at 20 ℃. In Queensland, soil temperature can reach below 13 ℃ between June and September (http://www.bom.gov.au/climate/australia/cities/). There is a possibility of higher TN loading into Queensland waters during these months. In addition, it is reasonable to assume that during this period NH_4^+ dominates TN resulting in increased environmental impacts. However, detailed scientific evidence of the influence of soil temperature on TN loading into groundwater and nitrogen speciation in the treatment chamber effluent for Queensland is currently not available.

b) OWTS density

Density refers to the number of OWTS per unit area. Guidelines for suitable lot sizes and densities of septic systems have been established as given in Table 3.10. Most of these studies consider pathogen dispersal when recommending lot sizes and system density and only limited attention is given to TN dispersal and use of treatment chambers other than septic tanks. System density is important because wastewater TN that is not removed or immobilised by individual systems cumulatively ended up in the groundwater (Dawes, 2005). The lack of consideration of TN loading when determining guidelines on lot sizes and OWTS density can pose a contamination risk to the receiving environment.

Table 3.10 Minimum sustainable lot sizes and OWTS density (Adapted from Beal et al., 2005)

Lot Size (m^2)	System Density ($/km^2$)	Comments	References
65,000	15	To prevent potential groundwater contamination	USEPA (1977)
2,000 – 4,000	250-500	To prevent increased TN below the soil absorption system	Perkin (1984)
1,000-12,000	85-1000	To prevent groundwater contamination	Yates and Yates (1989)
10,000	100	NSW guidelines for coastal towns based on minimum assimilative buffer area	Jelliffe (1999)
2,000 – 4,000	250-500	Based on average nutrient and hydraulic loading from OWTS and minimum setback distances to reduce pathogen risk	Geary and Gardner (1998)
50,000 – 100,000	10 -20	Recommended for environmentally sensitive areas based on N and P assimilation	Gerritse (2002)

c) Flow paths

Optimum treatment can be achieved through the movement of the wastewater through the biomat (Schwager & Boller, 1997; Van Cuyk et al., 2001). However, preferential flow paths may exist in a soil absorption system under certain conditions. Studies summarised in Table 3.11 lists the conditions where sidewall infiltration that by-passes the biomat will occur. Sidewall infiltration can reduce TN removal.

Table 3.11 Hydraulic flow behaviour in the soil absorption system

Sidewall infiltration conditions	References
Higher for sandy soils than finer-grained soils	McGauhey and Winneberger (1964)
Higher than bottom flow in some duplex soils	Brouwer et al. (1979)
Preferential pathway in clay loam soils during wet weather periods	Brouwer et al. (1979)
Flow rates ranging from 0 (non-permeable soils) to 0.09 m/day (permeable soils)	Bouma (1975)
Low lateral flows during wet weather due to low hydraulic gradient	Sherlock et al. (2002)

3.4.4 Soil characteristics

TN removal in the subsurface occurs as wastewater infiltrates and percolates through the soil. This may be influenced by the characteristics of the soil including pH, texture (clay content), structure, moisture content and temperature.

a) pH and alkalinity

Soil pH can directly influence TN removal in soil absorption systems. Nitrification, the first step in the TN removal process in the soil absorption system, favours alkaline conditions as the process consumes alkalinity. To nitrify, one gram of NH_4^+ requires 7.14 g of alkalinity measured as $CaCO_3$. In a laboratory experiment, Sahrawat (1982) found NO_3^- is only produced with a pH greater than 6. In the study, Sahrawat (1982) measured the formation of nitrate in 10 different soil samples after incubating them in an aerobic conical flask for 4 weeks at 30 ℃. According to the result of the study given in Table 3.12, TN in acidic soils did not nitrify under given conditions. The second step in TN removal in soil absorption systems is denitrification which converts NO_3^- to N_2. This process consumes acidity and hence occurs under acidic (low pH) conditions (Eq 3.11).

Table 3.12 Correlation between soil pH and TN concentration from the study by Sahrawat (1982)

Soil types used	pH	Total N (mg/kg)	NO_3 (mg/kg)
Calalahan sandy loam	3.4	1100	0
Malinao loamy sand	3.7	900	0
Luisiana clay	4.4	1750	0
Morong peat	5.6	5600	5
Lam Aw peat	6.1	12000	116
Maahas clay	6.5	1200	106
Quingua silty loam	6.5	1150	115
Pila clay	7.5	1850	123
Lipa loam	7.5	1900	98
Maahas clay, alkalised	8.6	1200	118

While calcareous sands such as in the sand islands on Queensland's coastline are favourable for nitrification due to high alkalinity, limited denitrification under these conditions may result in nitrate leaching into the groundwater. Despite favourable conditions, in acidic soils such as acid sulphate soils commonly found in Queensland, NO_3^- may act as a limiting factor (due to limited nitrification) for denitrification, thereby reducing the overall TN removal. Limited ammonia adsorption owing to the positive surface charge of acidic soils may also contribute to this phenomenon. Even in non-acidic soils, acidity can build in long-operating soil absorption systems due to nitrification, making conditions unfavourable for TN removal. However, studies into the impact of soil pH on OWTS TN removal and vice versa in Queensland soils are limited. This knowledge gap constrains the assessment of nutrient risk to the environment from OWTS.

b) Soil texture

Several studies have investigated the role of soil texture (particle size distribution of clay, silt and sand) on TN removal in a soil absorption system (Cogger & Carlile, 1984; Humphrey et al., 2010; Karathanasis et al., 2006). Soil texture is linked to the long-term acceptance rate of wastewater. It also indirectly influences the treatment process by controlling the hydraulic retention time (permeability rate). High infiltration rates occur in well-drained sandy soils. A high infiltration rate means low retention time which leads to limited denitrification (Meynendonckx et al., 2006; Mueller et al., 1997). Slow wastewater movement in fine-textured soils (e.g. clay loam) results in longer travel time, thus high denitrification. However, nitrification in poorly drained, fine-grained soils can be limited by low oxygen availability.

In the absence of detailed investigations into the variation in denitrification rates with soil type in relation to OWTS, literature available for different land use types was evaluated. A detailed investigation into denitrification rates in forest soils using different indicators including denitrification enzyme activity at the regional scale by Groffman and Tiedje (1989) provides strong evidence for the relationship between soil texture and denitrification rates. The study was conducted using nine soil types with varying percentages of sand, and drainage index, which is an index measure to represent plant available water content in soil under normal climatic conditions. According to the results presented in Table 3.13, the study found a strong negative correlation ($r^2 = 0.75$) between denitrification rates and the percentage of sand. Despite the percentage of sand, well-drained fine-textured soils also had low denitrification rates. Together, sand percentage and drainage index contribute to 86% of the variation in denitrification activity at a regional scale. These strong correlations can be attributed to the ability of sand percentage and drainage to control soil moisture content and oxygen availability that directly impacts denitrification. In addition, limited NO_3^- concentration due to reduced nitrification rates can also be attributed to the low denitrification rates in sandy soils. Groffman and Tiedje (1989) found low nitrogen mineralisation in sandy soils. In the context of OWTS, limited denitrification and mineralisation mean higher TN loading into the groundwater. Past studies have reported elevated nitrate concentrations in groundwater downstream of OWTS in sandy soils (Humphrey et al., 2010; Robertson et al., 1991). These findings indicate that OWTS in coastal areas with sandy soils can contribute to nitrogen contamination in receiving waters.

Table 3.13 Correlation between annual denitrification TN loss and soil texture and drainage index from the study by Groffman and Tiedje (1989)

Soil	Sand %	Drainage index	Denitrification (kg N/ha/year)
Loam			
Well drained	26	40	10
Somewhat poorly drained	42	50	11
Poorly drained	18	70	24
Clay Loam			
Well drained	34	44	18
Somewhat poorly drained	35	64	17
Poorly drained	15	74	40
Sand			
Well drained	76	40	0.6
Somewhat poorly drained	74	60	0.8
Poorly drained	77	61	0.5

Sexstone et al. (1985) investigated the variation in denitrification rates in sandy loam and clay loam soils after episodes of irrigation and rainfall using laboratory and field investigations. Increased denitrification rates were observed in sandy loam and clay soils (209 ng N/g/day and 303 ng N/g/day, respectively), with the increase in soil moisture due to rainfall or irrigation. The denitrification rates returned to pre-rainfall or irrigation levels for sandy loam (32 ng N/g/day) and clay soils (20 ng N/g/day) after 12 hrs and 60 hrs of the wetting events, respectively. Slow infiltration rates through fine-textured soils explain the difference in the lag period for the increase in denitrification rates and longer high nitrification rates after the wetting events (Sexstone et al., 1985).

However, application techniques and concentration of constituents in wastewater and runoff from other land use types are different, such as fertiliser application for example. Therefore, the applicability of nitrification-denitrification rates for other land use types compared to OWTS is questionable. Further research into nitrification-denitrification rates in soil absorption systems is needed.

c) Unsaturated zone thickness

The thickness of the unsaturated zone can significantly influence TN removal in the soil absorption system by influencing soil water content, aeration status, media surface area and hydraulic retention time. In general, wastewater treatment processes take place closer to the soil surface. Bunnell et al. (1999) investigated 11 standard septic systems in sandy soil in the Coastal Pinelands region, New Jersey, USA. The findings from this study given in Table 3.14 show a clear decrease in TN, NH_4^+ and organic N concentrations, and an increase in NO_2^- and NO_3^- concentrations with the depth. The study found minimal TN removal in soil absorption with nearly 34 mg/L (out of 54 mg/L) reaching the groundwater. Considering TN, about 34% was removed by the biomat and a negligible amount (4 mg/L) was removed in the deeper layers of the system due to denitrification (Bunnell et al., 1999). The researchers attributed the low denitrification rates to the oxic sand layer in the investigated system which could have inhibited nitrification. Further, TN removal by the biomat may vary with its characteristics. Therefore, the denitrification rates reported by Bunnell et al. (1999) may not be generic for all soil absorption systems.

Table 3.14 Characterisation of nitrogen species and their concentration in the soil absorption system from the study by Bunnell et al. (1999)

Sampling Zone	TN (mg/L)	NO_2^- + NO_3^-	NH_4^+	Org N
		as % of total N		
Septic Tank	**63.3**	**<1**	**78.7**	**21.6**
Top unsaturated zone (20 cm below the distribution pipes)	**41.5**	**80.2**	**12.3**	**7.5**
Top unsaturated zone (150 cm below the distribution pipes)	**34.0**	**94.4**	**2.1**	**3.5**

Most studies report on TN, NH_4^+ and NO_3^- concentrations in the wastewater and groundwater below a soil absorption system, but not within the soil absorption system. This limits the understanding of the TN removal process within the system. The comprehensive study by Bunnell et al. (1999) provides an in-depth insight into the TN removal processes and most importantly, identifies denitrification as not being a significant process in TN removal in sandy soils as expected. However, further detailed investigations to determine TN removal processes and their efficiencies in the soil absorption system will benefit in managing nutrient contamination risks from OWTS.

3.5 Research Outcomes on TN behaviour in Soil Absorption Systems

Table 3.15 Summarises the key research outcomes from selected studies into TN behaviours in soil absorption systems.

Table 3.15 Findings on TN removal in soil absorption systems (Adapted from Beal et al., 2004)

Reference	Study summary	Key findings
Starr and Sawhney (1980)	Wastewater TN and C migration in soil absorption system consisting of coarse sand.	Rainfall influenced nitrification. During the wet period, NH_4^+ travelled to deeper (> 90 cm) layers of the soil absorption system.
Kristiansen (1981)	TN removal by sand filters and the role of the biomat zone.	Low TN removal due to lack of labile C and anaerobic conditions.
Whelan and Barrow (1984)	Nutrient migration to groundwater from OWTS in sandy soil.	Organic N accumulated in the biomat zone. NO_3^- reached groundwater.
Cogger (1988)	Influence of loading rate and groundwater depth on nutrient migration to groundwater from septic systems in sandy soil.	High TN removal under unsaturated conditions. NH_4^+ reached groundwater under saturated conditions.
Pell and Nyberg (1989)	Measured nutrient and organic matter removal rates in newly operating sand filters and columns.	Total nitrification achieved in less than 15 cm of the sand filter depth.
Al-Shiekh Khalil et al. (2004)	TN removal from septic tank effluent using undisturbed soil (clay and sand) columns at the laboratory scale	Low TN removal in sandy soils compared to clay soil.
Geary (2005)	Use of piezometers and suction lysimeters to monitor groundwater and trace flow paths near a soil absorption system.	NO_3^- concentration was high near the soil absorption system and reduced with distance.
Bedessem et al. (2005)	TN removal in soil columns containing organic layers.	31% - 67% average TN removal in the columns with organic layers.

3.6 Summary of Key Findings

On-site wastewater treatment systems are considered a major source of nitrogen to both fresh and marine waters worldwide. Key reasons for this include high failure rates and limited TN removal in OWTS. In addition, the sensitivity of TN removal processes to various factors, such as groundwater level fluctuations, as well as the high mobility of nitrate in the subsurface, may also contribute to high TN leaching from OWTS.

System characteristics significantly influence the wastewater treatment processes and nitrogen removal and migration. Depending on the treatment chamber type (aerobic/anaerobic), the breakdown of complex organic compounds into basic organic compounds, such as amino acids to complete the hydrolysis to nitrate, may occur in the treatment chamber. Wastewater and OWTS characteristics influence the treatment chamber processes, which in turn can affect TN removal in wastewater disposal systems. For example, high organic carbon removal in the treatment chamber can negatively impact heterotrophic nitrification and denitrification in the subsurface. Therefore, investigations to establish a relationship between treatment chamber processes and TN removal merits future research. Further, considerations are currently given only to pathogen removal in OWTS design guidelines such as recommendations for buffer distances. The effectiveness of these design guidelines in preventing nitrogen contamination needs further research.

Several processes have been identified as contributing to TN removal in OWTS. This includes nitrification, denitrification, adsorption, ammonification, plant uptake and anammox are among the mechanisms in the subsurface. Nitrification and denitrification are identified as the key TN removal processes in soil absorption systems. However, the efficiency of these processes in OWTS-TN removal under Queensland conditions has received limited research attention and merits future research. Adsorption and anammox as potential TN removal mechanisms in soil absorption systems have gained limited attention, and are unlikely to be significant processes in many OWTS. Plant uptake is well understood as the key TN removal process in surface irrigation. It is worth exploring the possibility of using plant uptake as a potential mechanism to remove TN in soil absorption systems as well.

Detailed investigations into the fate and transport of TN in the subsurface are available in relation to fertiliser application. However, the applicability of this knowledge for nitrogen dispersal in OWTS is questionable. This is because of the differences in nitrogen speciation, dosing approach and infiltration rates in fertiliser application compared to wastewater disposal. In addition, the presence of a biomat in the soil absorption system may contribute to this difference. The biomat can regulate hydraulic conductivity and strain organic matter which supports TN removal processes. However, age and resistance of the biomat, and organic matter retention may not only negatively affect the TN concentration, but also the nitrogen speciation of the wastewater within the soil profile. However, an in-depth understanding of the contribution of biomat in TN removal is lacking at present.

Among wastewater characteristics that influence the TN removal process in the subsurface are, concentration of DO, Na^+ and organic matter. Site characteristics including the rainfall regime, temperature, OWTS density and preferential flow paths may also negatively affect TN removal in the subsurface. Texture, chemistry, pH and unsaturated zone thickness are the influential soil characteristics. The influence of these wastewater, site and soil characteristics in TN removal is reasonably well understood in many parts of the world. Although OWTS is considered the main domestic wastewater treatment option in some locations in Queensland, its efficacy in TN removal and the potential influence of wastewater, site and soil characteristics on TN removal is poorly understood and merits future research.

Key factors that are likely to decrease nitrogen removal for OWTS include high hydraulic conductivity and limited organic matter availability. These conditions are particularly present in coastal areas of Queensland. Based on the information available, it is likely that OWTS are not removing significant amounts of nitrogen and therefore nitrogen is migrating into the environment. The information available, despite current knowledge gaps, suggests that this migration will be significant when located in sandy coastal areas where subsurface infiltration is rapid, and groundwater is relatively shallow, and could lead to potential nitrogen contamination in receiving waters, Tools and methods available for assessing nutrient removal in OWTS and quantifying the movement of nutrients into the environment are discussed in Chapter 5.

Chapter 4. Factors Influencing Total Phosphorus Dispersal from OWTS

4.1 Introduction

Phosphorus naturally occurs in several forms, such as phosphates adsorbed to soil and sediments, organic phosphate and orthophosphate ($H_2PO_4^-$, HPO_4^{2-}, PO_4^{3-}). Phosphorus is an essential major nutrient for both terrestrial and aquatic life. Among the different forms, orthophosphate is the only bioavailable form which is generally available in low concentrations. Therefore, phosphorus can be considered a growth-limiting nutrient in most freshwater ecosystems. However, high phosphorus influx into surface waterbodies can influence their trophic status. Table 4.1 provides a general guide to the trophic state of surface waters based on phosphorus availability.

Table 4.1 Trophic state of surface waters based on total phosphorus (TP) availability

TP concentration (mg/L)	Trophic state	References
< 0.01	Oligotrophic (limited productivity)	Carey and Rydin (2011); Carlson and Simpson (1996)
0.01- 0.03	Mesotrophic (Intermediate productivity)	Carey and Rydin (2011); Carlson and Simpson (1996)
> 0.03	Eutrophic (enriched with nutrients and enhanced algal growth)	Dillon and Rigler (1974); Carey and Rydin (2011); Carlson and Simpson (1996)

The link between eutrophication and high total phosphorus (TP) availability in waterways is reasonably well understood. In an extensive review of long-term studies on aquatic ecosystems in Europe and North America, Schindler et al. (2016) noted that eutrophication can be controlled only by reducing the phosphorus input. The Peel-Harvey estuary in Western Australia has suffered several eutrophication events since 1980 due to high TP concentration (Williams, 2009). In 2006, when the images in Figure 4.1 showing the eutrophication status of Lake Goegrup, Peel-Harvey catchment were taken, the median TP concentration was 4.8 mg/L (Kelsey et al., 2011). This value was well above the phosphorus concentration required to trigger eutrophication (Table 4.1). Septic systems in the catchment were estimated to load nearly 34,000 kg/year of TP into the Peel-Harvey estuary and to be one of the key TP sources in the catchment (Williams, 2009). TP loading to Queensland waters from OWTS is yet to be understood and this gap constrains efforts to protect their trophic status and related ecosystem values.

Figure 4.1 Eutrophication of Lake Goegrup in Lower Serpentine River, Peel-Harvey Catchment (Kelsey et al., 2011)

Sources of phosphorus in the environment include rock weathering, mineralising organic matter, animal waste, fertiliser, sewage and industrial wastewater. On-site wastewater treatment systems (OWTS) have gained increasing attention in recent years as potential sources of phosphorus inputs to the environment. For example, it has been estimated that OWTS contributed 12% of anthropogenic phosphorus inputs to the Baltic Sea, Sweden (Boesch et al., 2006; Brandt & McManus, 2009).

Ek et al. (2011) estimated that 2.4×10^5 kg/year of TP was loaded into the Swedish marine environment from 7×10^5 OWTS compared to 6×10^4 kg/year loaded by 822 small municipal wastewater treatment plants (200 EP – 2000 EP) and 3.6×10^5 kg/year loaded by 476 large municipal wastewater treatment plants (> 2000 EP). TP loading from septic tanks contributed 2.6% (1.6×10^3 kg) of TP influx to the Peel-Harvey estuary, Western Australia, between 2006 and 2015 (Hennig et al., 2021). These study findings signify the importance of OWTS as a source of TP to aquatic environments.

This chapter provides insights into the TP removal processes in OWTS. It discusses the different TP removal mechanisms in the soil absorption system. The influence of wastewater characteristics, soil characteristics and other environmental factors on TP removal is also discussed.

4.2 Total Phosphorus Removal in OWTS

4.2.1 Total phosphorus removal in the treatment chamber

Total phosphorus (TP) removal in OWTS occurs in the treatment chamber and the soil absorption system. Mamals et al. (1994) reported that TP could precipitate with magnesium and ammonia in the wastewater to form struvite ($MgNH_4PO_4 \cdot 6H_2O$), based on studies undertaken at the San Francisco Southeast Water Pollution Control Plant, USA. However, evidence of struvite formations in OWTS is not available to date. Zanini et al. (1998) discussed the possibility of TP precipitating with iron in wastewater to form vivianite ($Fe_3(PO_4)_2.8H_2O$) in septic tanks. In aerobic treatment chambers, where oxidising conditions exist, iron is highly likely to be oxidised to Fe^{3+}. Under such conditions, the formation of vivianite is highly unlikely. Although the above research indicated precipitation as a possible TP removal mechanism, past research studies presented in Table 4.2 suggest that no measurable TP removal occur in the treatment chamber.

Table 4.2 TP removal in different types of treatment chambers

Type of Treatment Chamber	TP Concentration (mg/L)		References
	Influent	Effluent	
Septic Tank	5 - 20	5.2 – 17.0	USEPA (1998)
Septic Tank	3.2 – 6.6	3.1	Nasr and Mikhaeil (2013)
UASB-septic tank reactor	13.64	13.0 – 13.53	Al-Shayah and Mahmoud (2008)
Baffle septic tank	3.2 – 6.6	2.98	Nasr and Mikhaeil (2013)

4.2.2 Total phosphorus immobilisation in soil absorption systems

The TP removal in a soil absorption system varies widely (12% - 97%) (Carroll et al., 2006; Eveborn et al., 2014; Robertson, 2008; Robertson et al., 2012). This large variation in TP removal by soil treatment is attributed to soil characteristics, including soil texture, chemistry, and pH. Site characteristics such as the depth of the groundwater table, proximity to surface water sources as well as climate may also be influential. OWTS characteristics such as age and usage (seasonal/regular) may also influence TP removal efficacy.

Partially treated sewage dispersed to a soil absorption system can have a TP concentration between 3.1 – 17.0

mg/L (USEPA, 2002). In the environment, TP does not have a gaseous phase and follows a sedimentation-type cycle (Schlesinger & Bernhardt, 2013). This means TP that is not subjected to plant uptake is either adsorbed or precipitated in soil. A portion of the TP load can also remain in dissolved form in the soil solution and leach down potentially reaching the groundwater. TP can subsequently enter surface waters when groundwater is discharged to surface waters. Figure 4.2 provides a schematic diagram of the phosphorus cycle.

Adsorption to soil and sediment particles is an important TP immobilisation mechanism in soil and groundwater. This is particularly true in Australia as the soils often have very low phosphorus levels. TP can be adsorbed to soil particles by forming surface complexes with aluminium, iron (hydro) oxide, and carbonates (Robertson, 2008; Spiteri et al., 2007; Zurawsky et al., 2004). Precipitation is another key TP removal mechanism in a soil absorption system (Robertson et al., 2012). In iron and aluminium-rich soils, phosphorus precipitation reactions are more prominent compared to phosphorus adsorption (Zanini et al., 1998). TP can precipitate as aluminium (Al^{3+}; Eq 4.1) and iron (Fe^{3+}; Eq 4.2) phosphate and, to a lesser extent as calcium (Ca^{2+}; Eq 4.3) phosphate (Eveborn et al., 2012; Robertson et al., 2012). TP retention in soils increases more or less proportionately to the Al^{3+} and Fe^{3+} concentrations and pH in the soil. Using microprobe images of sand used in the filter bed of a domestic OWTS located in Parry Sound, Ontario, Robertson et al. (2012) found ferric and aluminium oxyhydroxide mineral precipitate on the surface of sand grains (Figure 4.3). This finding suggests that precipitation is the major TP removal mechanism in medium-coarse sand filter beds.

$$Al^{3+} + H_nPO_4^{3-n} \leftrightarrow AlPO_4 + nH^+ \quad \text{(Eq 4.1)}$$

$$Fe^{3+} + H_nPO_4^{3-n} \leftrightarrow FePO_4 + nH^+ \quad \text{(Eq 4.2)}$$

$$10\,Ca^{2+} + 6\,PO_4^{3-} + 2\,OH^- \leftrightarrow Ca_{10}(PO_4).6(OH)_2 \downarrow \quad \text{(Eq 4.3)}$$

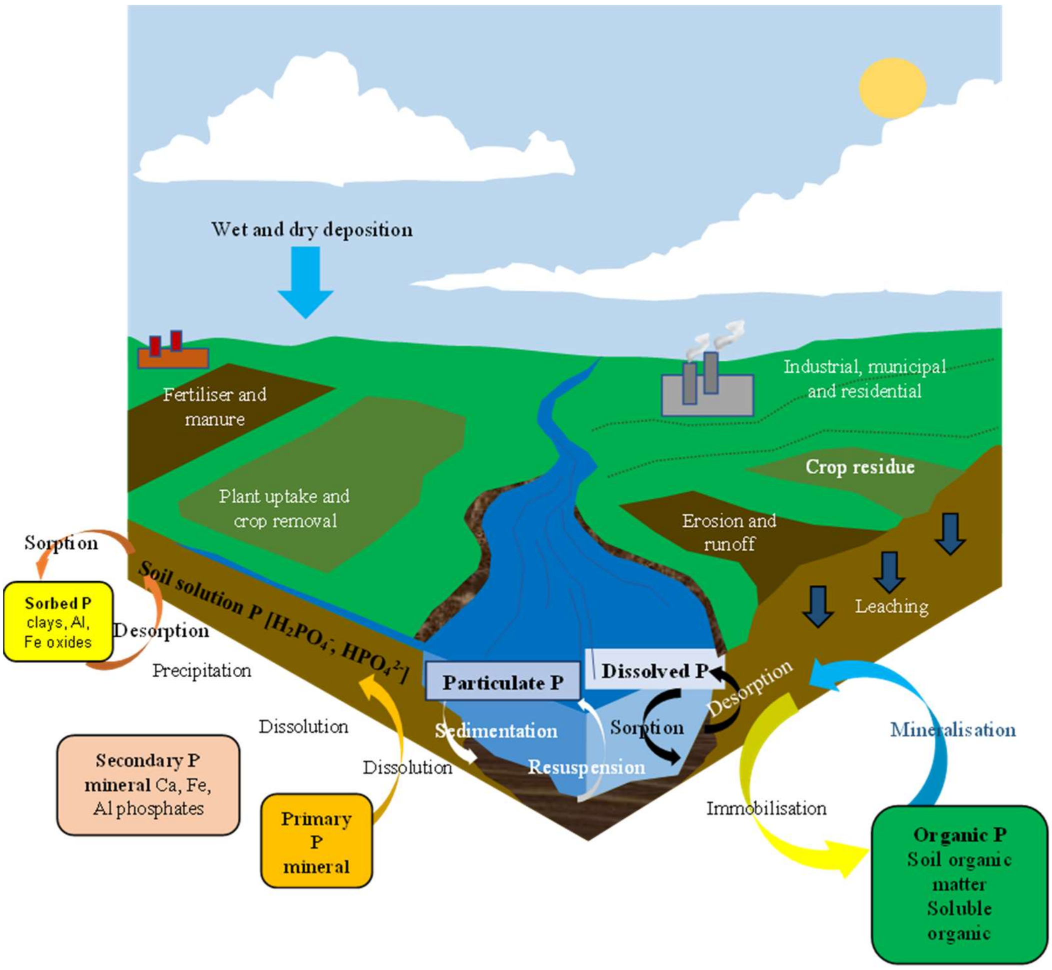


Figure 4.2 Phosphorus Cycle

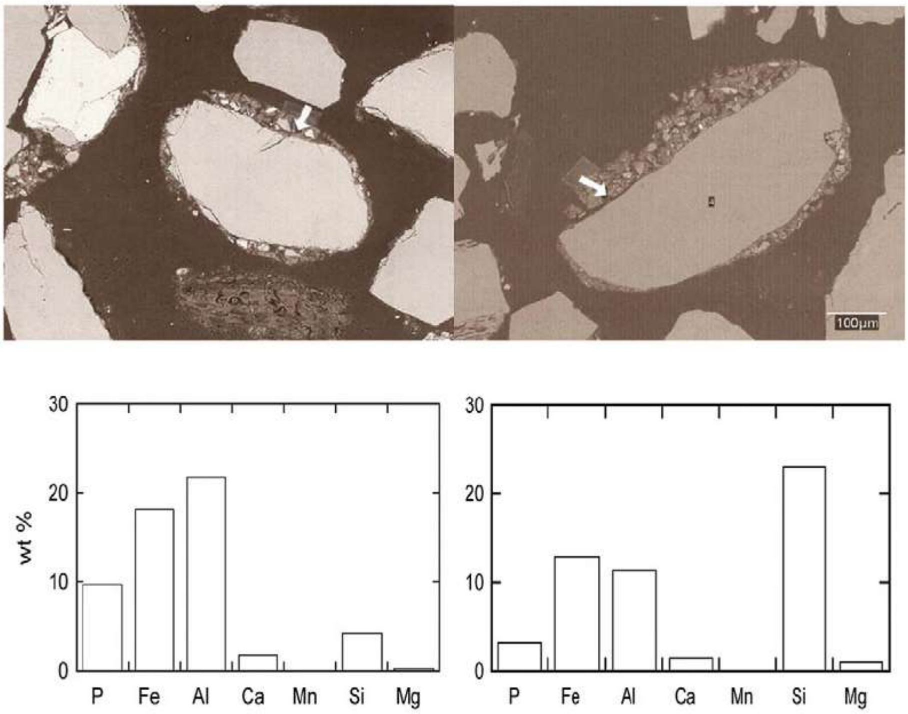


Figure 4.3 Backscattered electronic images of quartz grains from a sand filter with secondary mineral coatings. The histograms give the elemental composition of the mineral coating (Robertson, 2012)

4.2.3 Total phosphorus immobilisation in surface irrigation

The TP in partially treated wastewater dispersed using surface irrigation generally infiltrates into the soil. Infiltrated TP migrates through the soil profile before reaching ground or surface water. The fate and transport of TP in the subsurface are discussed in detail in Section 4.2.2. Gerritse (2002) calculated the migration time of TP using an empirical formula at two different reaction rates (b1 and b2, Eq 4.4) in different soils in Western Australia. The study outcomes are given in Table 4.3. In this study, all wastewater TP was assumed to infiltrate at a rate of 100 mm/day. High wastewater TP concentration (23 mg/L) was used in the study acknowledging the high evaporation enrichment in the region. However, this study did not account for the TP loss due to plant uptake. While similar studies are rare, the applicability of this data to Queensland is questionable. This is not only because of the significant difference in soil properties but also because of the climatic conditions. Accordingly, representative data for soil density, adsorption properties and wastewater concentration need to be used for calculating TP migration rates in Queensland.

$$t_{1\%} = (A\, C_m^{\,(b1-1)}\, \rho\, L\, D^{-1}\, 365)^{1/(1-b2)} \qquad \text{(Eq 4.4)}$$

Where:

- A - adsorption coefficient
- ρ - bulk density of soil or aquifer
- b1 - adsorption parameter, describes the decrease in K_d with increasing solution concentration of TP.
- b2 - reaction parameter, describing the increase in K_d with time
- D - discharge
- L - travel distance
- $t_{1\%}$ - the time it takes for the phosphate concentration at a particular depth in the soil to increase by 1% of the input concentration
- C_m - mg P in a litre of soil solution

Table 4.3 Calculated vertical migration time of phosphate (3.5 kg P/year) dispersal in an irrigation field of 150 m^2 through a 0.5 m thick soil with a density of 1.6 kg/L (from the study by Gerritse 2002)

Soil type	Location	PRI (mg/kg)	A	b1	b2	Infiltration rate (mm/day)	Distance to Groundwater (m)	P in the effluent (mg/L)	Travel time (year)
Grey sand	Ellen Brook	0.5	1	0.6	0.2	100	0.5	23	<1
Gavin sand	Harvey River	0.7	3	0.4	0.2	100	0.5	23	1
Karakatta sand	Baldivis	5	18	0.4	0.2	100	0.5	23	30
Yellow sand	Gnangara	7	25	0.4	0.3	100	0.5	23	25
Calcareous sand	Rottnest	12	35	0.4	0.25	100	0.5	23	190
Lateritic sandy loam	Mundaring	15	50	0.3	0.34	100	0.5	23	50
Lateritic loam	Mundaring	50	100	0.3	0.25	100	0.5	23	150
Lateritic loam	Mundaring	750	300	0.3	0.25	100	0.5	23	650

Note: PRI – phosphorus retention index, a measure of adsorption or retention capacity of soil for phosphate, measured in P mg/Kg. Refer to Bolland and Windsor (2007) for relating PRI to phosphorus buffering index (PBI) and Burkitt et al. (2002) for an explanation of phosphorus buffering capacity (PBC) and phosphorus buffering index (PRI).

4.2.4 Total phosphorus remobilisation in a wastewater disposal area

Total phosphorus removal in soil, commonly expressed in terms of the distribution coefficient (K_d; Eq 4.5) is in equilibrium with TP concentration in the soil solution. Precipitated TP can be dissolved if the concentration in the soil solution (C_m) reduces (Zanini et al., 1998). Under laboratory conditions, Lookman et al. (1994) and Zurawsky et al. (2004) observed TP leaching when C_m was reduced by loading deionised water into soil columns. Therefore, it is reasonable to assume that any process that dilutes the TP concentration in the soil solution, such as adding diluted wastewater and/or infiltrating rainwater, can result in TP leaching due to dissolution. TP leaching could also be influenced by the soil characteristics and when soil is saturated with TP, then the soil solution will have a higher portion of TP that can be leached down with rainfall or irrigation.

$$K_d = C_s / C_m \qquad \text{(Eq 4.5)}$$

Where:

K_d - distribution coefficient

C_s - the amount of phosphate adsorbed to a unit weight of soil (P mg/kg soil)

C_m - concentration of phosphate in the soil solution (P mg /L of soil solution)

In general, soils have high phosphorus storage capacity, and the first 2 m in soils such as vertosol can hold TP from secondary treated sewage (~15 mg/L of TP) for up to 100 years. However, the conditions in soil absorption systems are different to surface irrigation and therefore the phosphorus storage in soil absorption systems may vary accordingly. Lookman et al. (1994) found that the TP absorbed by aluminium oxide can readily desorb contributing to significant TP leaching.

Further, the reduction of Fe^{3+} to Fe^{2+} (reductive dissolution of iron phosphate) can also increase TP mobility in the subsurface (Peiffer and Wan, 2016; Zurawsky et al., 2004). The reductive dissolution can occur due to the reducing potential of partially treated wastewater and the anoxic conditions deep in the soil absorption system, close to the groundwater table. Alternative reducing and oxidising conditions can also occur in OWTS used seasonally, such as in holiday homes. Oxic conditions occur below the biomat due to its resistance as discussed in Section 3.3.1 when the OWTS is operational. During a long hold in operation (in the off-season), the biomat could become inactive due to the lack of nutrients for microbes. Even after the OWTS re-commences operation, it will take time for the biomat to re-establish. This can result in relatively high wastewater infiltration rates and lead to anoxic conditions below the biomat. Other processes that result in anoxic conditions and consequent reductive dissolution of phosphorus in the subsurface may include high wastewater loading rates and rainfall.

Fe^{2+} can also precipitate with phosphorus (Zanini et al., 1998). However, stoichiometrically 1 g of Fe^{2+} can precipitate only 1.1 g of phosphorus, whereas 1 g of Fe^{3+} can precipitate 1.7 g of phosphorus. Therefore, it is evident that the dissolution of iron can release phosphorus in the soil absorption system. Fluctuating groundwater table can also result in alternating oxidising and reducing conditions in the subsurface. Consequently, this can result in the dissolution of $FePO_4$. The effects of groundwater level fluctuation on phosphorus dissolution in OWTS constitute a knowledge gap that merit future investigations.

Dissolution and/or desorption of phosphorus may occur over the long term even after an OWTS has been decommissioned. For example, Robertson (2008) observed a phosphorus plume in groundwater downgradient to decommissioned septic systems in Langton and Long Point 2, Ontario. This may be because circumstances such as changes in redox conditions and the groundwater table can result in the dissolution and/or desorption of phosphorus. Therefore, the long-term viability of a soil absorption system as a sink for TP from wastewater treatment systems is questionable.

4.3 Influential Factors in TP Removal

TP removal in OWTS is governed by adsorption and precipitation mechanisms. These chemical reactions are influenced by several factors, such as wastewater characteristics, site characteristics and climate. Understanding the factors influencing TP removal is essential for assessing the treatment performance of OWTS. The discussion below provides an evaluation of the factors influencing TP removal in a soil absorption system.

4.3.1 Characteristics of wastewater

Characteristics of wastewater vary widely depending on the type of dwelling/source, the diet of the occupants, household detergents used as well as the waste stream. Variations in wastewater characteristics, including pH, total phosphorus concentration, major cation concentrations and the content of humic substances can directly influence phosphorus precipitation and adsorption in OWTS (USEPA, 2002). Table 4.4 summarises the average composition of domestic raw sewage constituents that can influence TP immobilisation in OWTS.

Table 4.4 Characteristics of raw sewage that can potentially influence TP removal

	Robertson (2012)	USEPA (2002)	Nasr and Mikhaeil (2013)
pH	6.8 ± 0.4	6 - 9	5.5 - 7.7
TP (mg/L)	na	6 -12	3.2 – 6.6
PO_4–P (mg/L)	7.5 ± 1.1	na	na
Ca (mg/L)	5.0 ± 2.3	na	na
Fe (mg/L)	0.31 ± 0.13	na	na
Al (mg/L)	0.34 ± 0.01	na	na
Mg (mg/L)	0.4 ± 0.23	na	na

na – not available

a) Total phosphorus concentration

In wastewater, TP is present in inorganic form. Table 4.2 gives the average TP concentration in the treatment chamber effluent which is discharged to the environment via wastewater disposal area. Depending on the concentration, TP removal mechanisms in a soil absorption system can change. The formation of calcium or iron phosphate precipitates only occurs at relatively high TP concentrations (Gustafsson et al., 2012). At concentrations below 5 mg/L, TP is generally adsorbed to the surface of iron and aluminium minerals (USEPA, 2002; Whelan, 1986). However, whether phosphorus adsorption occurs simultaneously with phosphorus precipitation is not explicitly known.

b) Presence of aluminium, iron and calcium

Domestic wastewater contains aluminium (Al), calcium (Ca) and iron (Fe) (Table 4.4). Although the exact mechanisms are yet to be understood, these cations are considered to play a key role in initial TP removal in the treatment chamber. TP immobilisation in the treatment chamber is important as it removes a certain amount of TP from the system with the settled sludge. Aluminium and iron in wastewater can also contribute to TP immobilisation via soil adsorption in OWTS. Robertson et al. (2012) found that 33 kg of phosphorus had precipitated in an iron and aluminium-poor soil absorption system at Parry Sound, Ontario over a period of 22 years. It was hypothesised that iron and aluminium in wastewater contribute to forming an iron and aluminium-rich authigenic mineral precipitate. Zanini et al. (1998) discussed the possibility of Fe in wastewater forming $Fe_3(PO_4)_2$ precipitate. The above studies confirm that the chemical composition of wastewater plays an important role in TP removal.

c) Wastewater loading rate

Hydraulic loading rates significantly impact the long-term TP removal performance of soil absorption systems. Under high hydraulic loading, the phosphorus sorption capacity of subsoil can be exceeded rapidly. Eveborn et al. (2014) found a significant increase in the time taken for phosphorus saturation with a commensurate reduction in the hydraulic loading rate. The increase in the time required for phosphorus saturation in soil was from 15 to 40 years when the hydraulic loading rate reduced from 6 mm/day to 3 mm/day. However, the study found that the time taken to reach saturation would change with site-specific factors such as clay content. A low hydraulic loading rate does not change phosphorus mechanisms or the TP removal capacity of soil absorption systems. Therefore, the observed increase in phosphorus storage time may only be a result of reduced TP loading or low P concentration in the treated sewage. In addition, higher loading rates can result in anaerobic conditions in the subsurface leading to high TP mobilisation as discussed in Section 4.2.3.

4.3.2 Site characteristics

a) Groundwater

Although not a part of a soil absorption system, groundwater can contribute to the TP removal process in OWTS. During a six-year groundwater monitoring program at Parry Sound, Central Ontario, Robertson et al. (2012) found that 3.1 mg/L of TP attenuated within a 5 m downgradient from a single home septic system. The study did not clarify the exact mechanisms responsible for TP reduction. However, TP attenuation in groundwater is less effective in the presence of calcareous sediments and karst aquifers (Robertson, 2003, 2008; Robertson and Harman, 1999). The low TP removal in the above lithologies can be attributed to the high hydraulic conductivity under the subsurface conditions discussed. Rapid wastewater mobility due to high hydraulic conductivity can reduce phosphorus precipitation. Therefore, soil texture, depth to groundwater, aquifer material and hydraulic conductivity must be considered in performance assessment. Accordingly, appropriate setback distance must be established based on the aquifer material and the different lithologies.

b) Distance to surface water resources

Distance to surface waters can influence TP immobilisation. Relatively longer distances to surface water sources can ensure prolonged contact with soil particles, resulting in relatively higher TP removal (USEPA, 2002). OWTS close to surface waters will increase the risk of phosphorus pollution due to possible leakages from the treatment system (Robertson, 2008). Figure 4.4 illustrates the migration of a phosphorus plume from an OWTS to a surface water body. The concentration of phosphorus in the plume reduces as it travels away from the treatment system. This indicates that the lateral distance to surface water is an important factor in TP removal.

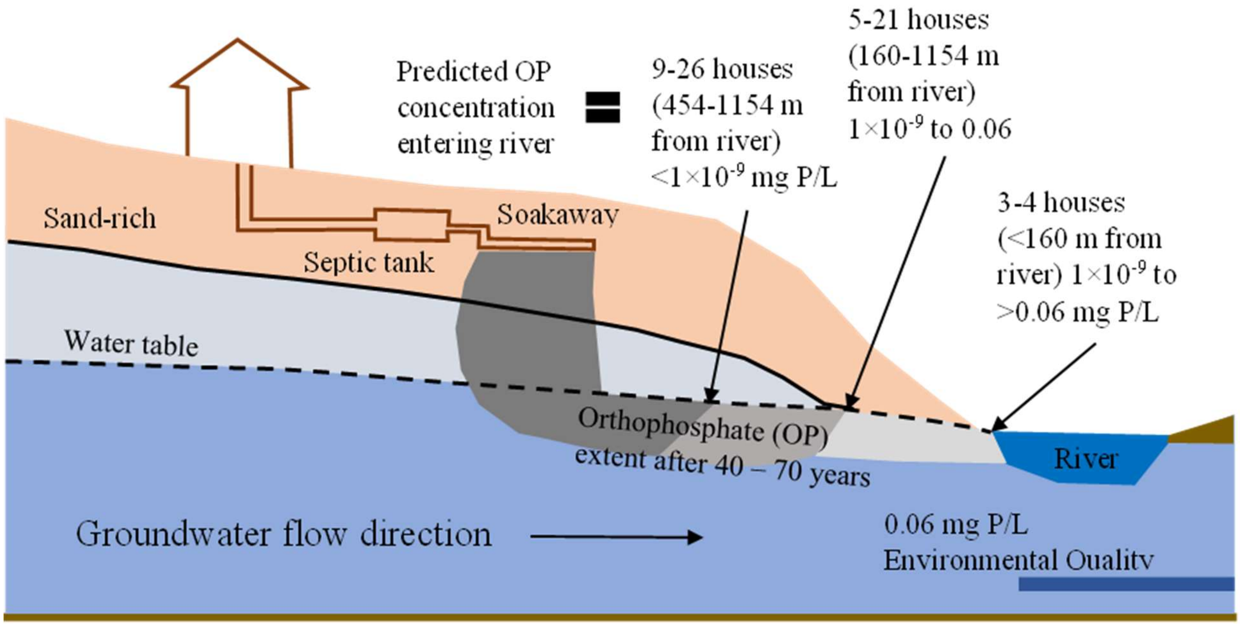


Figure 4.4 Migration of a phosphorus plume in Upper River Nar, UK (Adapted from Speed et al., 2019)

c) Climate

Climate, particularly rainfall plays a significant role in TP removal. For example, Robertson (2012) observed a temporal increase in groundwater TP concentration downgradient to a Parry Sound septic system in Ontario, Canada. This was attributed to the dissolution of immobilised TP in infiltrating water resulting from snow melting. Eveborn et al. (2014) also observed TP leaching with the addition of distilled water to soil columns obtained from different soil absorption systems. The dissolution of immobilised TP in the soil solution under highly saturated conditions can be attributed to high TP leaching as observed in the above studies. Similarly, reductive dissolution may also contribute to high TP leaching under wet climatic conditions. A discussion on the leaching of TP under wet climatic conditions is provided in Section 4.2.3.

The influence of climatic conditions in TP immobilisation/mobilisation in the subsurface was evidenced in the study undertaken at the Johnstone River Catchment, Queensland (Hunter and Walton, 2008). The total phosphorus influx from non-agricultural areas of the Johnstone River basin was found to be two to three times greater than from the agricultural areas in the upper part of the catchment (Hunter and Walton, 2008). The significant differences in the influx highlight the influence of rainfall on subsurface phosphorus mobility (Table 4.5). The main phosphorus source in the upper Johnstone River is iron-rich basaltic rocks (McCulloch et al., 2003). However, the contribution of OWTS as a potential source of phosphorus and the temporal variations in TP loading from OWTS is largely unknown in Queensland and merits future research.

Table 4.5 The role of catchment characteristics on phosphorus mobility in the Johnstone River Catchment, Queensland (source: Hunter and Walton, 2008)

Factor	Upper Johnstone River Catchment	Lower Johnstone River Catchment
Soil/Rock Type	Red ferrosols, red dermosols, red kandosols	Brown dermosols and redox hydrosols
Rainfall (mm/year)	1673	3545
Phosphorus influx from non-agricultural lands (kg/ha/year)	0.8 ± 0.1	2.3 ± 1.9

d) OWTS density

High OWTS density will increase TP input into the subsurface. The soil absorption systems may not be able to effectively remove TP if the input is higher than the sorption capacity of the soil. Similarly, high OWTS density can increase the amount of TP leaching into groundwater due to the relatively larger volume of wastewater dispersed collectively. This can affect the TP immobilisation process in groundwater systems. Unfortunately, there is very limited knowledge available on the cumulative impacts of OWTS density on TP immobilisation in both, soil and groundwater systems. This is a significant knowledge gap and merits future research.

4.3.3 Soil characteristics

a) Phosphorus Buffering Capacity (PBC)

The soil's ability to 'lock up' phosphorus is referred to as the Phosphorus Buffering Capacity (PBC). Phosphorus sorbed into the soil is in equilibrium with the dissolved phases in the soil solution. Soils can buffer any changes to the phosphorus concentration in the soil solution due to the addition or removal of phosphorus and this ability is defined as PBC. PBC is related to the availability of iron, aluminium and calcium compounds in the soil. In a study conducted using 290 samples, Burkitt et al. (2002) found that the PBC of Australian surface soils varies widely between 0 – 218 mg P/kg. The study reported higher PBC (218 mg P/kg) in ferrosol from the wet tropics of north Queensland whereas the sandy soils had very low values (0 – 4 mg P/kg). Soils with low PBC are favourable for agriculture due to their limited capacity to bond with the phosphorus in the fertiliser applied making the plant available phosphorus amount high. In contrast, soils with high PBC would be favourable for soil disposal systems. However, there is a paucity of knowledge on the impact of PBC on wastewater TP removal.

b) Soil texture and hydraulic conductivity

Total phosphorus removal in soil absorption systems is highly dependent on the texture (clay content) of the soil. Fine textured soils without continuous macropores enhance TP removal by reducing the wastewater percolation rate (USEPA, 2002). Slow percolation increases the interaction between soil and wastewater, leading to relatively higher TP removal. The high surface area or large volume of fine-textured soils can also contribute to enhancing TP removal due to increased P storage (sorption) capacity. The depth of the unsaturated zone also enhances TP removal by increasing wastewater contact with the soil. However, Robertson and Harman (1999) identified a considerable difference in TP mobility in different septic tanks located in similar textured soil. It is hypothesised that this is because the soil texture alone does not influence TP removal and that a range of other factors such as initial P concentration, pH, bulk density and K_{sat} can also play a role.

c) pH

TP removal in soil absorption systems can be significantly influenced by the soil pH as evident from the outcomes of studies summarised in Table 4.6. Under acidic conditions, adsorption is the dominant TP removal mechanism, whereas, under alkaline conditions, precipitation is dominant. Acidic conditions can also increase TP removal by leaching iron and aluminium in the soil (Gustafsson et al., 2012; Robertson et al., 2021; Zanini et al., 1998). Overall, acidic soils such as acid sulphate soils are favourable for TP removal. However, detailed investigations on TP immobilisation in OWTS systems in Queensland are lacking.

Wastewater inputs, mainly quality, can influence soil pH in soil absorption systems. For example, in a detailed investigation using soil columns from different soil absorption systems, Eveborn et al. (2014) observed a 1 to 2 point decrease in soil pH with the introduction of wastewater. The decrease in pH results from organic matter degradation and nitrification of NH_3 (Robertson, 2003). Therefore, with wastewater addition, TP removal efficacy in soil may increase. However, calcareous soils can buffer the pH changes resulting from wastewater oxidation. Therefore, wastewater addition may not necessarily influence TP removal in calcium-rich soils compared to sandy soils in coastal regions.

Table 4.6 TP removal mechanism in a soil absorption system under different pH conditions

pH	Removal mechanism	References
<6.0	Adsorption to Fe^{2+}, Fe^{3+} and Al^{3+} and/or precipitation of iron or aluminium phosphates (strengite, variscite)	Vohla et al. (2011); Vymazal (2004); Gustafsson et al. (2012); Eveborn et al. (2014).
~7.0	Adsorption to carbonates	Sø et al. (2011)
>9.0	Precipitation of calcium phosphate	Gustafsson et al. (2012); Johansson (1999)

d) Soil chemistry

Soil chemistry such as cation exchange capacity (CEC) is one of the most important factors influencing TP removal in soil absorption systems. Soil chemical characteristics can influence adsorption and precipitation mechanisms by providing the required cations and controlling the soil pH. Iron, aluminium, and calcium are major cations responsible for forming bonds with TP. Hydrated iron or aluminium oxides form monodentate and bi-nuclear complexes with phosphorus (Vymazal, 2004). Iron, aluminium and calcium can also immobilise TP by forming precipitates in soil. For example, Lookman et al. (1994) observed high TP removal in aluminium-rich sandy soils, whilst Robertson and Garda (2020) observed an oxidised pyrite (FeS_2) deposit in the subsurface preventing the migration of TP at an OWTS at Long Point, Ontario.

Despite having a higher concentration of calcium ions, calcareous soils are relatively less effective in TP removal (Robertson, 2008, 2012; Robertson and Harman, 1999). Calcium-rich soils can buffer soil pH with the addition of wastewater, making conditions unfavourable for aluminium and iron mineral precipitation. Another reason would be the low solubility constant of calcium phosphate (1.3×10^{-32}). Due to the low solubility constant, calcium phosphate precipitate can dissolve in water. This explains the migration of TP plumes even in the case of decommissioned systems located in calcareous soil (Robertson and Harman, 1999). The influence of soil calcium concentration on TP removal is of particular importance in the coastal regions of Queensland and merits future research.

e) Organic matter content

Soil organic matter includes low molecular weight organic acids (LMOA), humic and fulvic acids, and organic matter leachate. Only a limited number of studies have investigated the influence of organic matter on TP removal, and the outcomes reported are relatively inconclusive. Column studies undertaken have reported a negative relationship between organic matter content and TP removal (Alvarez et al., 2004; Bahl and Pasricha, 1998; Hue et al., 1994; Lookman et al., 1994; Nziguheba et al., 1998; Song et al., 2006). The outcomes from these studies have attributed reduced TP adsorption to competition between organic matter and TP for sorption sites in soil which is termed competitive inhibition. However, in an extensive review of published literature, Guppy et al. (2005) suggested that competitive inhibition is only significant in the rootzone (rhizosphere) of the soil and may not be relevant to OWTS.

In contrast, some past studies have found that organic matter can increase TP removal efficacy in soils. For example, Eveborn et al. (2014) noted that organic matter can increase TP removal by absorbing TP. However, the longevity of organic matter as a possible sink for TP is questionable. This is because organic matter can subsequently decompose due to microbial action and TP can be released back into the environment.

However, there is limited knowledge of the impact of soil physicochemical characteristics on TP removal in OWTS. A possible reason for the paucity of knowledge would be the difficulty in simulating soil physical properties in column studies at the laboratory scale. Another reason would be the difficulty in conducting field studies and sampling to investigate TP mobility in the subsurface. Based on the literature available, it appears that only one research group has undertaken in-depth field studies on TP immobilization in soil absorption systems. This work was based on a few selected sites in Ontario, Canada (Robertson, 2012; Robertson and Harman, 1999; Zanini et al., 1998). Under the circumstances, this important area that needs further research.

4.4 Enhancing Total Phosphorus Removal in OWTS

Wastewater TP removal in the treatment chamber is minimal and is mostly dispersed to the environment via a soil absorption system (USEPA, 2002). Considering the growing interest and demand for TP removal in OWTS, several approaches have been considered to enhance TP removal capability. These different approaches can be categorized as follows:

1. Source Separation
2. Chemical precipitation
3. Reactive filters

However, these technologies may have significant disadvantages, which limit their practical application. Some of the disadvantages include increased maintenance, the need for frequent desludging and urine removal, which will result in logistical challenges and an appreciable increase in costs.

4.5 Summary of Key Findings

Amongst nutrients in wastewater, TP removal by OWTS has received limited attention. Reasons for this are attributed to the relatively high TP sorption capacity of soils and a limited understanding of the environmental impacts of TP. The environmental impacts of TP from OWTS cannot be underestimated. This is because TP can influence the trophic state of surface waters, particularly freshwater. Therefore, OWTS as a potential point source of TP contamination requires further investigation.

In the case of OWTS, partially treated wastewater containing phosphorus is dispersed into the soil for further treatment and the soil characteristics govern the removal process. The role of soil chemistry and pH on TP removal is reasonably well understood. However, TP immobilisation in soil absorption systems is not necessarily a sustainable process. In other words, although TP immobilisation in the subsurface will occur initially, soils may only have the capacity to do so for a certain number of years to decades. Dissolution and desorption of immobilised TP can occur under specific circumstances. Infiltrating rainwater, diluted wastewater and rising water table can re-mobilise the already immobilised (adsorbed/precipitated) TP in the subsurface. The TP load that can leach under these circumstances varies depending on soil chemical characteristics, immobilisation mechanism and the amount of pore water present in the soil or the soil moisture content.

Several approaches have been proposed to enhance the removal of TP discharged from OWTS. These approaches include source separation and precipitating TP using chemical compounds (chemical dosing) or reactive filters. However, these approaches which are discussed in Chapter 5 have inherent limitations and may not be commonly used in OWTS.

Chapter 5. Evaluation of Nutrient Removal Performance of OWTS

5.1 Introduction

On-site wastewater treatment systems (OWTS) release wastewater into the environment using disposal fields after primary treatment. Disposal fields can be either subsurface or surface infiltration and are expected to provide further treatment before being dispersed to the broader environment. In some reported instances, the disposal of inadequately treated wastewater containing nutrients and pathogens has resulted in significant environmental impacts. Poor design, maintenance and siting of OWTS can be considered a hazard and pose a contamination risk to the receiving environment, such as soil, groundwater or nearby surface waters. Such systems can also create a risk to human health. Therefore, the treatment performance and associated risks need to be better understood to help develop strategies to mitigate environmental and human health risks associated with the use of OWTS.

5.2 OWTS Failure

Identifying OWTS failures is important for mitigating risks associated with the use of OWTS. This includes providing recommendations on siting, designing and maintaining as well as regulating long-term impacts by performance monitoring. However, using currently available literature for guidance in formulating OWTS risk management strategies is not straightforward as there is no commonly accepted definition for system failure. The definitions provided vary widely with the objectives and the scope of the study. For example, there is a noticeable difference in failure rates reported in the different studies listed in Table 5.1 due to the variations in the failure definition used. The average OWTS failure rate in the USA is given as 10 - 20% (USEPA, 2002). However, states such as Georgia and Massachusetts, which consider public health hazards as the failure definition, report lower failure rates (1.7 and 25%, respectively). Other states such as Missouri and Minnesota, where hydraulic failure and ground and surface water contamination are considered report high failure rates (30 – 70%). Goonetilleke and Dawes (2001) and Goonetilleke et al. (2002) estimated 70 – 90% failure rates in Logan and Gold Coast City Council areas based on detailed investigations of treatment chamber performance, householder maintenance practices and effluent quality. Therefore, the lack of a clear definition of failure makes OWTS risk assessment challenging.

Table 5.1 OWTS failure rates

OWTS Failure rate	Failure definition	References
10 - 20% USA (on average)		USEPA (2002)
1.7% Georgia 25% Massachusetts	Public health hazards	Nelson et al. (1999)
30 – 50% Missouri 50 – 70% Minnesota	Hydraulic failures & Ground/surface water contamination	
50 – 90% in NSW (out of 284,000)	System or Hydraulic failures	Kenway and Irvine (2001)
70 – 90% in Brisbane, Logan and Gold Coast	System or hydraulic failures	Goonetilleke and Dawes (2001); Goonetilleke et al. (2002)

In the context of this review, any scenario that leads to the leaching or surcharging of nutrients above a defined threshold value into ground or surface water is considered a failure or causing potential environmental harm. Failure can be categorised as system failure or hydraulic failure. System failure is defined as the inability of the wastewater disposal area (subsurface or surface soil system) to treat the effluent to the required quality before being potentially discharged into the groundwater. Hydraulic failure is defined as the inability of the treatment chamber or the wastewater disposal area to accommodate the wastewater volume being discharged leading to effluent either ponding on the surface and/or surface flow being discharged off-site. It is important to note that the definitions of these two types of failures are not always exact and generally overlap. Both system and hydraulic failures can lead to the treatment failure, presenting a contamination risk to the receiving environment.

5.3 Risk Associated with Failed OWTS

In generic terms, risk can be described as the likelihood of a course of action(s) that will result in an event leading toward a potential for harm (hazard) (AS/NZS 4360:2004). In the context of OWTS, the release of inadequately treated wastewater can potentially harm human health and the surrounding environment. Direct human and environmental exposure to untreated wastewater may occur due to poor design, location and maintenance resulting in system failure leading to leakage, ponding and surfacing (Moriarty and Nokes, 2014). Once dispersed to the environment via a wastewater disposal area (subsurface/surface), nutrients and pathogens can harm the soil (the immediate receiving environment), groundwater and surface waters fed by groundwater. Indirect human exposure to pathogens and nutrients via contaminated groundwater and surface waters is also a possibility. Therefore, there is always human and environmental health risk associated with the use of OWTS.

5.4 Tools (Models) Available to Assess the Risk Associated with OWTS

OWTS risk assessment is essential to ensure the sustainability and non-polluting conditions of on-site wastewater treatment. A risk assessment framework can be used as a tool to guide the risk management process. The research tools available in this space are abundant, ranging from basic spreadsheet models to sophisticated empirical models that can assess the risk associated with OWTS. Some of these models, such as **SANICOSE (**Gill and Mockler, 2016) and the model by Withers et al. (2011) were developed specifically for OWTS. Other models, such as **TOUGHREACT-N** (Maggi et al., 2008) and **HYDRUS-2D (**Šimůnek et al., 1999) were developed for the assessment of other hazards but have been applied to assess OWTS. Models have taken a range of approaches for assessing the risk associated with OWTS as listed in Table 5.2. The three generic modelling approaches applied to risk assessment are:

1. OWTS vulnerability to failure approach
2. Ground/surface water vulnerability to contamination approach
3. Risk assessment approach

Figure 5.1 below schematically illustrates the scope and application of the above approaches in assessing the risk associated with the use of OWTS.

Table 5.2 Different approaches available to assess the risk associated with the use of OWTS

Approach	Examples
OWTS vulnerability to failure	• GIS-based framework for septic system vulnerability (Hoghooghi et al., 2021) • Land Use Risk Tool (LURT) (https://www.seqwater.com.au)
Ground/surface water vulnerability to contamination	• DRASTIC model (Aller et al., 1985) • WBLMER - NLM (Valiela et al., 1997) • ArcNLET (Zhu et al., 2016) • SANICOSE (Gill and Mockler, 2016) • STUMOD-FL-HPS - (Sawyer, 2015)
Risk assessment approach	• Integrated Risk assessment framework (Carroll et al., 2006)

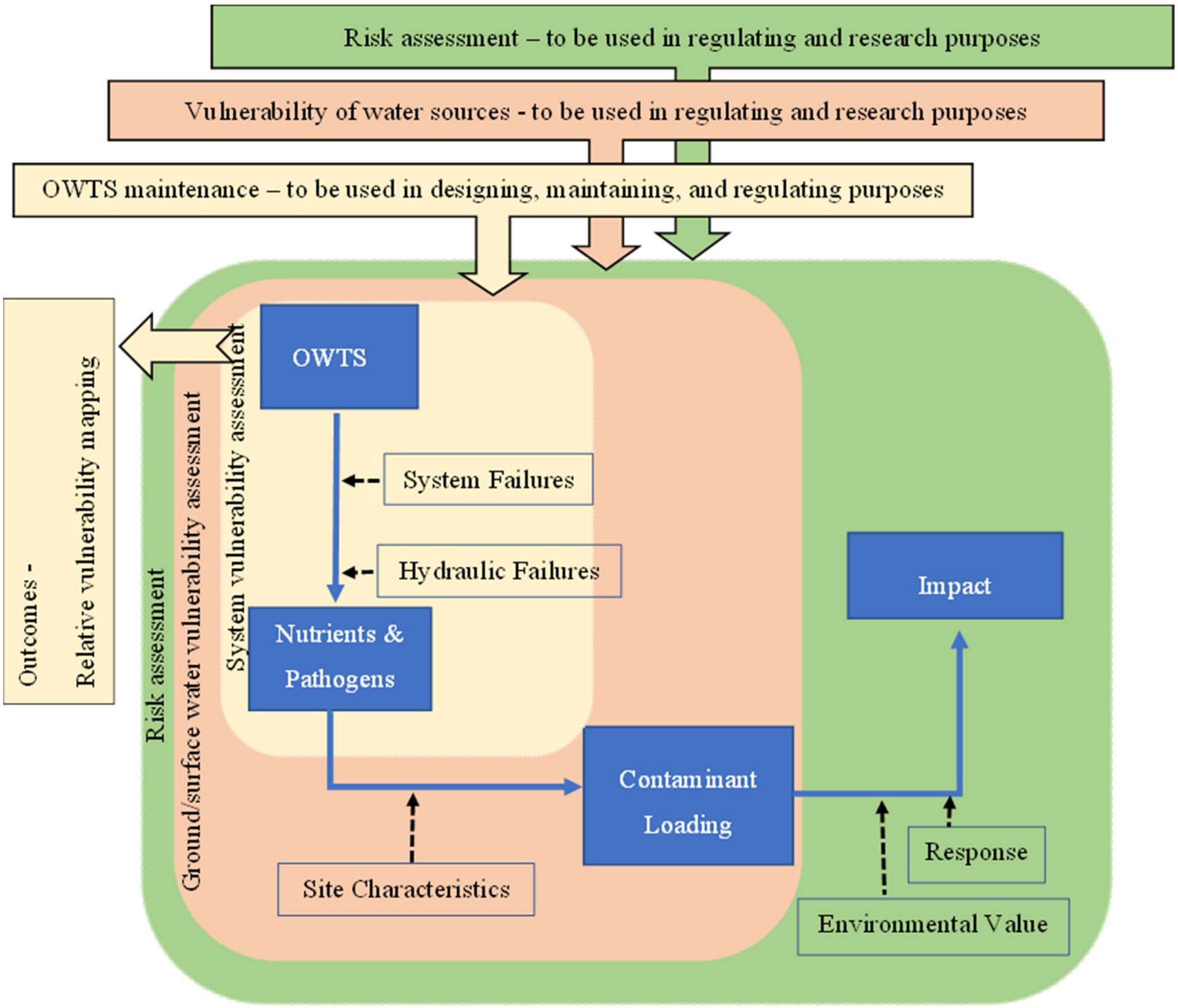


Figure 5.1 Schematic showing the key approaches adopted to assess OWTS performance.

5.4.1 Assessing OWTS vulnerability to failure

Vulnerability is defined as the degree to which human or environmental systems are likely to experience harm due to perturbation or stress (Popescu et al., 2008). OWTS vulnerability to failure generally considers any system or hydraulic failure that results from the characteristics of OWTS, including design, siting, construction and maintenance practices (Hoghooghi et al., 2021). Changes in environmental conditions, such as high rainfall events that result in hydraulic failure may also be considered (Kohler et al., 2016). Hoghooghi et al. (2021) adopted statistical frameworks to produce GIS maps showing replacement rates of 3,792 septic systems in the coastal plains of southern Bryan County, Georgia, USA. In this study, two well-known statistical frameworks, the 'conditional inference trees model' and the 'logistic regression model' were used to qualitatively analyse the OWTS replacement rates as a function of system characteristics and soil hydraulic characteristics. Using the generalised linear regression model, Kohler et al. (2016) assessed the degree to which a system loses functionality (fragility) due to hydraulic stresses in 225 septic systems in Boulder County, USA. The study modelled the frequency and severity of the failure based on resulting repairs as a function of rainfall, temperature and waterways. The waterways were considered as these integrate rainfall, soil moisture and watershed response and indicate the subsurface conditions. According to the study, 70% of the OWTS failures were related to a weather event, which included prolonged rainfall patterns, wetter-than-average months and high temperatures.

The OWTS vulnerability approach generally produces vulnerability maps at the local scale. The outcome may present as failure percentages or relative likelihood of failure to occur (high, medium, low). Outcomes of OWTS vulnerability assessments can be used in the planning and decision-making about new OWTS at a given location or region. More importantly, these types of assessments can be used to manage and regulate existing systems and to identify regions where potential environmental harm can occur and management measures such as water quality monitoring and OWTS inspections are required for early detection of failure (Hoghooghi et al., 2021). In addition, OWTS vulnerability assessments are important for homeowners to obtain a good understanding of possible failures and required maintenance during the life cycle of OWTS as well as the associated costs.

Among the drawbacks of OWTS vulnerability assessments is the extensive data requirements. Data such as system dimensions, vent arrangements and age may simply not be available for existing systems. Further, the OWTS vulnerability to failure approach does not consider the factors/subsurface characteristics that influence the fate and transport of nutrients in the subsurface such as nitrification/denitrification rates, assimilation, adsorption and desorption. Therefore, low OWTS vulnerability to failure does not necessarily mean that groundwater (and/or surface waters) may not be contaminated by nutrients from OWTS. Table 5.3 provides a summary of the characteristics of a selected list of models used to analyse OWTS vulnerability to failure.

Table 5.3 A summary of characteristics of models commonly used to assess OWTS vulnerability to failure.

Model	Overview	Reference
Land Use Risk Tool (LURT)	**Description:** An online tool that can be used for decision-making on new OWTS. **Capabilities:** • Can predict the relative contamination risk that can be posed by OWTS. • Helps identify sites that require detailed soil investigations before design and implementation. • Determines the specific design loading/irrigation rate based on soil permeability. **Limitations:** • Only assesses the contamination threat from OWTS in drinking water catchments. • Only considers widely used OWTS system designs or combinations. **Comments:** • Freely available, an easy-to-use tool. • The model can only provide a general assessment.	https://www.seqwater.com.au
GIS-based framework for septic system vulnerability	**Description:** Assessment is done using two regression analysis methods; the Conditional inference trees model and the Logistic regression model. **Capabilities:** • Able to assess septic tank failure due to groundwater level fluctuations. • Can be used to determine appropriate OWTS design parameters. **Limitations:** • Assumes OWTS failures only result from tank capacity and soil hydraulic properties. • Details of system characteristics are required. • The model focuses on septic tank failures that would lead to replacement and repair. **Comments:** The model does not consider the nutrient contamination risk posed by OWTS.	Hoghooghi et al. (2021)

5.4.2 Assessing groundwater vulnerability to OWTS contamination

Receiving environments are vulnerable to the impacts of OWTS effluent. The receiving environments include soil, the primary receiving environment, and groundwater, the secondary receiving environment. To the best of available knowledge, the impact on soil related to OWTS use has not been assessed in scientific literature. Therefore, this report only considers the groundwater vulnerability which is discussed below.

Groundwater vulnerability can be defined as 'an intrinsic property of a groundwater system that depends on the sensitivity of that system to human and/or natural impacts' (Liggett and Talwar, 2009). Groundwater vulnerability assessment accounts for the specific vulnerability of a given region. This includes the physical characteristics of the environment (intrinsic vulnerability to contamination) and the transport properties of nutrients in the subsurface (Liggett and Talwar, 2009). Once dispersed into the disposal area (surface or subsurface), partially treated wastewater infiltrates and percolates through the unsaturated zone potentially reaching the groundwater. The nutrient load reaching the groundwater and the time taken is a function of OWTS characteristics as well as the site characteristics that influence nutrient reduction and transport. These factors include soil characteristics (texture, CEC, phosphorus buffering capacity and hydraulic conductivity), unsaturated zone thickness (depth to groundwater), and biological activity such as the presence of tree/crop roots.

User-friendly simple models to data-intensive complex models have been developed to integrate the influential factors needed to assess groundwater contamination vulnerability. These models generally produce maps that rank relative vulnerability (high, medium, low) or estimate nutrient loading. TOUGHREACT-N (Maggi et al., 2008), DRASTIC (Aller et al., 1985) and HYDRUS-2D (Šimůnek et al., 1999) are groundwater vulnerability assessment models. Whilst most models have been developed to assess groundwater vulnerability to various contaminants introduced at the land surface such as fertiliser, road salt, chemical spills, and landfill leachate (Liggett and Talwar, 2009), a few models such as **WBLMER – NLM** (Valiela et al., 1997) have been specifically developed for OWTS. However, models that were originally developed to investigate contaminants applied on the land surface may also be adapted to assess OWTS systems. A detailed description of selected models that can be used to assess groundwater vulnerability to nutrients from OWTS is provided in Table 5.4.

All the models that are used to assess the vulnerability of groundwater require input parameters such as OWTS characteristics, wastewater characteristics, site characteristics (soil type, texture, hydraulic properties, depth to groundwater and slope) and site-specific climate data. These models may vary in terms of complexity, scale, scope, intended outcomes (qualitative or quantitative), resources and data requirements (Focazio et al., 2002). Based on the method used, Liggett and Talwar (2009) have categorised vulnerability assessment into three types.

1. Index (and overlay) method
2. Statistical method
3. Process method

a) Index method

The index method determines the relative risk posed by contaminants at the regional/catchment scale (Liggett and Talwar, 2009). This is a popular approach due to its modest data requirements and easy implementation. Broad risk categories (low, medium, high) are the outcome of an index approach that can incorporate GIS technology to produce low-cost, user-friendly regional scale risk maps. These maps are essential tools for decision-making on OWTS such as determining the level of site assessment required and identifying regions that require relatively more frequent monitoring (Liggett and Talwar, 2009) or requirement for a centralised system.

The approach includes assessing the overall 'pollution potential' or 'hazard' of OWTS-sourced nutrients and combining it with the possible adverse impacts on the receiving environment. Estimating hazards requires understanding the factors influencing nutrient surcharge or leaching from OWTS. Known as 'hazard indicators' factors such as loading rate, soil type, rainfall, operational practices and hydraulic conductivity may be included. A risk level (low, medium, high) or a subjective numerical value is assigned (e.g. 1 to 5 scale) to hazard indicators. The risk level of each hazard indicator is then combined to estimate the overall hazard. The approach does not require an understanding, with certainty, of how these hazard indicators cause failure (Liggett and Talwar, 2009). Figure 5.2 provides an overview of a hazard assessment using the DRASTIC index model.

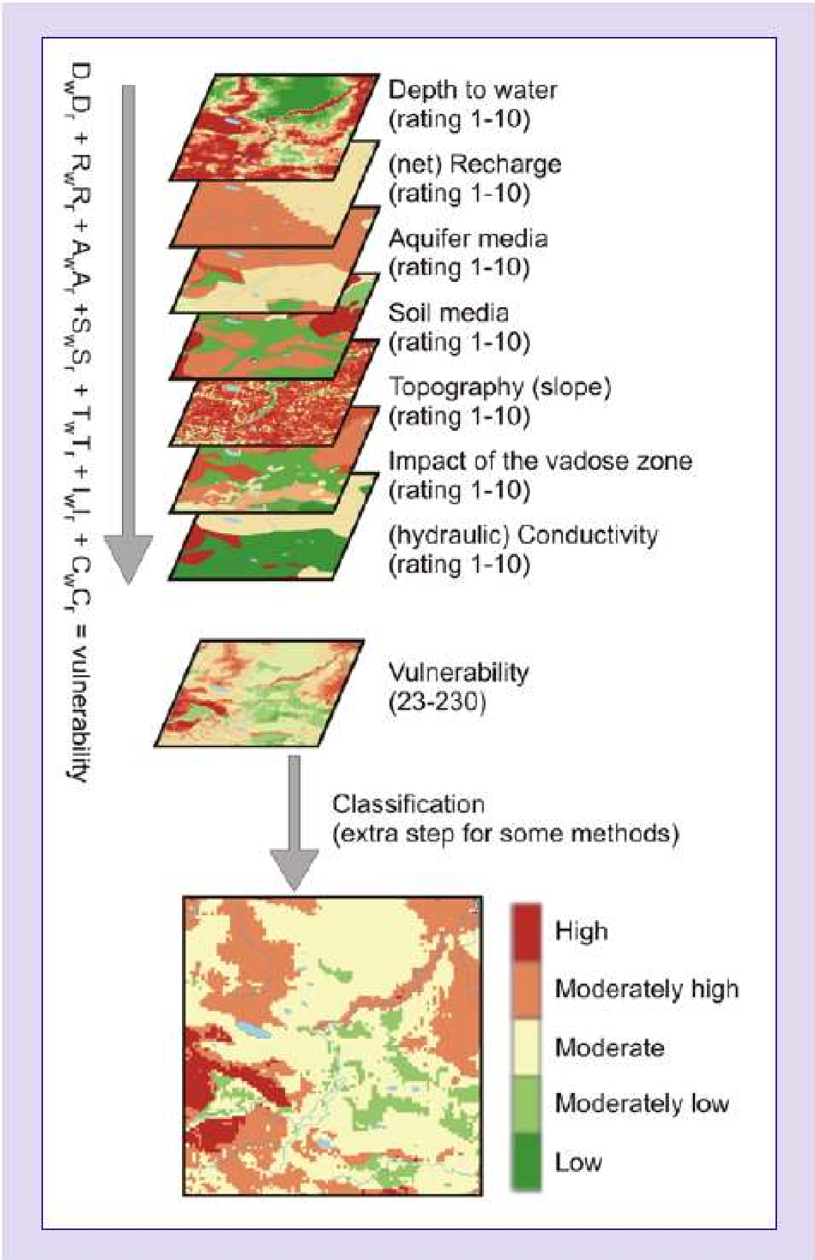


Figure 5.2 The application of DRASTIC model to assess groundwater vulnerability to contamination (Liggett and Talwar, 2009)

b) Statistical method

Statistical models can assess the vulnerability of groundwater to contamination by determining the probability of nutrients from OWTS exceeding a certain concentration or threshold level. The method requires estimating the initial nutrient loading from OWTS and measuring water quality values in the receiving environment. Based on the mass balance principle, linear regression equations can be developed for relating the nutrient loss during migration to site characteristics. The method requires a relatively high amount of data compared to the index method and is more appropriate for vulnerability assessment over a large area (Liggett and Talwar, 2009). Despite an extensive review of the literature, this study was unable to find a linear regression equation that could explain the fate and transport of nutrients from OWTS.

c) Process method

The process method, such as **WBLMER – NLM** and ArcNLET listed in Table 5.4, utilises a deterministic approach for assessing groundwater vulnerability. The assessment may include estimating travel time, concentrations, and duration of contamination using analytical solutions or numerical computer models. The method may include unsaturated zone processes, and/or saturated zone processes. This type of data-intensive, sophisticated method requires extensive resources and involves significant costs and expertise. In addition, the process method may only be appropriate to assess the vulnerability at the local (point) scale. However, this method is widely used for assessing the vulnerability of groundwater to contamination in general and specifically from OWTS. The ability to produce quantitative estimates is another advantage of the process method.

Table 5.4 Summary of commonly used models for assessing ground and surface water vulnerability to nitrogen contamination from OWTS

Model	Overview	Reference
	Models based on the index method	
DRASTIC	**Description:** A sophisticated GIS-based model to assess groundwater vulnerability to contamination. **Capabilities:** The model can be enhanced by either adding or substituting parameters depending on the site characteristics. **Limitations:** • Only applicable to the contaminants that are discharged/applied on the ground surface. • Limited to contaminant migration resulting from infiltrating rainwater. • Only limited applicability in small lots due to low resolution (0.4 km^2). **Comments:** • The model can be adapted to assess groundwater vulnerability to contamination from surface irrigation but may require major modifications to assess soil absorption systems. • Model output must incorporate scientific observations of the area for use in the risk assessment.	**Developed**: Aller et al. (1985)
	Models based on the statistical method	
The model developed by Withers et al. (2011)	**Description**: A model to assess the generic contamination risk from OWTS at the catchment scale. **Capabilities:** Estimates annual contaminant loading including heavy metals and other chemical compounds and electrical conductivity into surface waters. **Limitations:** • Only considers contaminant loading into surface water sources. • Does not consider the influence of site characteristics on contaminant loading. **Comments**: May not be suitable for assessing the nitrogen contamination risk of OWTS as the subsurface nitrate processes are not considered.	**Developed**: Withers et al. (2011)

Model	Overview	Reference
Models based on the process method		
WBLMER – NLM Nitrogen loading model	**Description**: An empirical model that estimates the TN loading into receiving waters and provides insight into the sources, losses, and transport of TN in groundwater moving through a coastal watershed. **Capabilities:** • Able to estimate TN loading from different sources including OWTS. • Able to calculate TN losses due to subsurface processes. **Limitations:** • Suitable for rural to suburban catchments underlain by unconsolidated sandy sediments. **Comments**: • Could apply to the coastal regions in Queensland with local input data. • Could be highly sensitive to TN loading from OWTS as the model considers all soil absorption system processes in estimating TN losses.	**Developed**: Valiela et al. (1997)
The model developed by Singh et al. (2017)	**Description:** A simple catchment scale hydrogeology-based model to assess TN attenuation in the subsurface and to predict TN loading to surface waters from OWTS. **Capabilities:** Considers the impact of biological growth and soil and rock interactions for determining TN losses. **Limitations:** • A generic model that considers monthly TN loading into a waterway from different sources including OWTS. • Does not identify the source. • Does not consider the impact of climate on TN attenuation. • Does not estimate TN loading into groundwater. **Comments:** The model over-simplifies nitrogen processes in the subsurface which may compromise the accuracy of the output.	**Developed**: Singh et al. (2017)

Model	Overview	Reference
ArcNLET ArcGIS-based Nitrate Load Estimation Toolkit	**Description:** A GIS-embedded numerical model for flow and transport simulation to estimate TN loading into receiving waters from OWTS. **Capabilities:** • Able to simulate the transport and transformation of nitrate and ammonium in the vadose zone. • Considers groundwater nitrate and ammonia, transport with advection, dispersion and denitrification. **Limitations:** • Uses empirical nitrification and denitrification rates instead of redox conditions which may compromise the accuracy.	**Developed**: Zhu et al. (2016)
SANICOSE – Source Apportionment of Nutrients in Irish Catchments for On-Site Effluent	**Description:** Semi-quantitative/qualitative decision support tool using mass balance approach to predict the annual TN and TP loading to surface waters from OWTS. **Capabilities:** • Determination of ammonium and nitrate concentrations in the vadose zone and groundwater. • Outputs can be integrated with other tools such as the Source Load Apportionment Model (SLAM) that has been developed for catchment nutrient characterisation. **Limitations:** • Assumes that the groundwater profile is a replica of the topography. • Not able to model temporal variations in TN transport, particularly after influential events such as heavy rainfall. **Comments:** • Useful from a pollution management perspective as the model can predict the flow paths. • Easy-to-use tool for estimating long-term TN loading into surface water sources with moderate data requirements.	**Developed:** Gill and Mockler (2016)
STUMOD-FL-HPS	**Description:** Combined aquifer and soil model to predict contaminant fate and transport adapted to Florida conditions. **Capabilities:** • Ability to use default parameters as well as user-specified inputs allowing model calibration/validation to site-specific data. • The degree of soil treatment, fate and transport in groundwater and quantitative estimations of N removal can be obtained as outputs.	STUMOD-FL-HPS (2015)

Model	Overview	Reference
	• Can account for plant nutrient uptake. • Able to incorporate limitations in denitrification with carbon availability. • Ability to account for the cumulative impacts of multiple OWTS. **Limitations:** • Unable to adequately predict the performance of all possible OWTS design and operation configurations. • Steady-state, one-dimensional flow conditions are assumed (variable operating or environmental conditions are not considered). • Does not account for transformations of organic nitrogen. • Assumes effluent application over the infiltration surface and similar infiltration rates across the infiltration surface. • Does not account for varying soil and groundwater characteristics. **Comments:** • Simple-to-use, robust, spreadsheet-based tool to evaluate TN fate and transport from OWTS under a range of operation and site conditions.	

5.4.3 Integrated risk assessment approach

Risk is a function of hazard, the likelihood of a hazardous event (vulnerability) and the consequences of the hazardous event (Liggett and Talwar, 2009). In the context of this review, any failure of OWTS that leads to nutrient discharge off-site can be considered a hazard. The likelihood of a hazardous event occurring (vulnerability) is dependent on a range of system and receiving environment characteristics. The consequences of such a hazardous event are dependent on the environmental values of the receiving environment.

Risk = f (vulnerability, hazard, consequence) (Liggett and Talwar, 2009)

Risk assessment furthers the vulnerability assessments and considers the risk posed by OWTS. Carroll et al. (2006) proposed an integrated risk assessment model using multivariate analytical methods based on a case study area in the Gold Coast region to assess human and environmental health risks from OWTS. However, the focus was microbial pathogen contamination of receiving waters. It is possible that this model could be adapted for other regions to assess the risk of nutrient contamination. One of the limitations of this model is that it does not account for varying boundary conditions (e.g. variation in groundwater table) and climatic conditions.

5.5 Summary of Key Findings

The lack of a clear definition of OWTS failure makes it difficult to derive conclusions from the available literature on OWTS performance. In this review, any scenario that leads to leaching or surcharging nutrients above a defined threshold value is considered a treatment failure. Evaluating performance and associated risks relating to OWTS usage is essential for implementing effective strategies to mitigate wider environmental and human health implications or consequences (risks). The broad attention to assessing the risk associated with OWTS is evidenced by the number of risk assessment models available.

These risk assessment models can be broadly categorised into three groups based on the assessment approach. The first is the OWTS vulnerability to failure approach, which assesses any treatment failure that results from the characteristics of OWTS, including design, siting, construction and maintenance practices. The second is the groundwater vulnerability to contamination approach which considers the sensitivity of the groundwater system to be impacted by human and/or natural hazards. Third, some models further the vulnerability assessments and consider the overall risk posed by OWTS. Risk is a function of hazard, the likelihood of a hazardous event (vulnerability) and the consequences of the hazardous event. The few available OWTS risk assessment models generally focus on assessing the public health risk rather than the nutrient contamination risk.

In summary, assessing the performance of OWTS and associated environmental and human health impacts is an important need. The approaches and methods adopted in OWTS risk assessments are indirect and vary widely with the scope of the study. Further, irrespective of the similarities in the circumstances, risk assessment models need to be adapted to the local conditions and qualified to be used in a given setting. STUMOD-FL-HPS for example is a simple spreadsheet model developed to quantify TN transport through unsaturated zone to the groundwater in Florida, USA. Using analytical solutions, the model determines the water movement and TN transport considering soil texture, layers, hydraulic parameters and nitrification-denitrification rates. This model could be adapted using local conditions in estimating TN loading into Queensland waters. Further work is needed to test and refine available tools or develop new tools to reliably assess the potential environmental risk associated with OWTS, specifically in relation to nutrients and potential impacts on environmental values.

Chapter 6. Conclusions

On-site wastewater treatment systems (OWTS) are commonly used in Australia in rural and regional areas and peri-urban fringes of major urban centres where reticulated sewerage systems are not available. Available information suggests that OWTS often only involve primary treatment and are not adequately maintained/serviced, resulting in limited, or poor, nutrient (and pathogens) removal. Also, there is evidence of common system failure which increases with age. Factors influencing the treatment performance of OWTS, and environmental impacts associated with poorly performing OWTS are well documented around the world.

Accordingly, it can be concluded that OWTS pose a significant risk to environmental values, including aquatic ecosystems and human health, particularly in coastal areas with shallow groundwater and sandy substrates. Despite this potential risk, significant knowledge gaps exist in relation to OWTS nutrient removal performance and the fate of wastewater nutrients and pathogens. As a result, robust tools are not available to help assess the potential risks, leading to challenges for improvements in the design, management and regulation of OWTS to minimise these risks. Conclusions around the potential impacts of OWTS, factors affecting TN and TP removal and methods for evaluating nutrient removal are discussed below.

6.1 Potential Impacts of OWTS

Extensive investigations undertaken in Australia and overseas have identified OWTS as a significant source of nutrients in groundwater as well as surface waters including oceans. Nutrients can significantly impact the environmental values of groundwater as a drinking water source. Studies in the USA have reported levels above drinking water guideline values for nutrient enrichment in groundwater as a result of wastewater disposal from OWTS. The trophic status of surface waters can also be impacted by nutrients originating from OWTS. Studies in Australia have established a direct link between nutrients from OWTS and the eutrophication of surface waters. Despite being a potential source of nutrients to the aquatic environment in Queensland, investigations into OWTS have received only limited attention.

In addition to nutrients, pathogens can also impact the environmental values of receiving waters. Disease outbreaks related to OWTS have been reported in Australia and around the world. In addition to human health impacts, recent studies have shown coral health can also be impacted by human enteric pathogens. Therefore, appropriate siting and investigations into the effectiveness of current buffer distances in preventing the degradation of sensitive aquatic ecosystem values are needed, in addition to considering drinking and recreational values.

The introduction of wastewater can change the microbiome in the soil absorption system and the soil below it. The growth of a biomat below the infiltration surface is one of the discernible and well investigated changes to soil microbiome, in particular for short term wastewater application. However, the long-term impact of wastewater application on the soil ecosystem and changes to the ecosystem deeper in the soil absorption system are yet to be investigated. Moreover, OWTS as a potential contributor to greenhouse gases also merits future research.

6.2 Factors Influencing TN Dispersal

Significant knowledge gaps exist in relation to TN removal and dispersal from OWTS. Based on the general wastewater treatment concepts, the treatment chamber of an OWTS, which typically undertakes primary treatment only, should remove a significant amount of organic matter and pathogens. Similarly, depending on the oxygen availability, the composition of nitrogen species could be significantly modified by biological processes, but the overall TN concentration should remain relatively unchanged. However, detailed studies on TN treatment and removal in the OWTS treatment chamber are limited.

Processes including nitrification, denitrification, adsorption and plant uptake can immobilise TN in the soil environment. Considerable knowledge of these processes is available in relation to fertiliser application. However, the difference in concentration and composition of nitrogen species, dosing mechanisms and infiltration rates between fertiliser usage and wastewater disposal limits the application of this knowledge to OWTS.

TN removal in the wastewater disposal area can be influenced by factors including wastewater, site and soil characteristics. In soil absorption systems, the biomat can also influence TN removal. Although studies have been undertaken in other countries such as the USA, and other states of Australia, knowledge of TN removal processes and the potential impact of the influential factors is lacking for Queensland. This is an important area for further research.

6.3 Factors Influencing TP Dispersal

OWTS as a potential source of TP has received limited attention, leading to significant knowledge gaps and significant challenges in undertaking meaningful risk assessment. Although the formation of low soluble phosphate minerals is a possibility, the understanding of TP removal in the treatment chamber is limited. However, intuitively, if solids built up in the treatment chamber are not removed regularly, the long-term removal of TP in the treatment chamber will be limited. Although yet to be established, the biomat is likely to contribute to TP removal.

The immobilisation of TP in the soil absorption system is well understood. The primary TP immobilisation mechanisms include adsorption and precipitation which are influenced by soil chemistry. However, the impact of organic matter content on TP immobilisation is little understood and requires further investigation. Although the TP adsorption capacity of soils is considerably high, desorption and dissolution may occur under certain circumstances such as high infiltration events and changes in the redox potential. Therefore, the reliability of soil as a sorbent in immobilising wastewater TP in the long term is questionable. In Queensland, where distinct dry and wet seasons exist, changes in pore water TP concentration and redox potential leading to TP leaching are a possibility. Therefore, investigations on TP immobilisation in soil absorption systems in Queensland soils merit further research.

6.4 Performance Evaluation of Nutrient Removal

The absence of a clear definition of OWTS failure makes deriving conclusions on system performance challenging. This study has identified scenarios that can lead to the leaching or surcharging of nutrients above a defined threshold value as treatment failure. A number of models are available to assess the risk posed by OWTS. These models adopt a range of approaches and methodologies based on the scope, resource availability and the need of the investigations. Further, irrespective of the similarities in the circumstances, risk assessment models need to be adapted to the local conditions and checked for suitability for a given setting.

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Appendix A: Sewage-associated impacts on coral

Impacts on coral due to the increase in nutrient concentration in ambient water

There are four major impacts on coral due to increased nutrient concentrations in ambient seawater as identified in the research literature. The impacts include: (1) algae growth; (2) coral disease and bleaching; (3) coral growth changes; and (4) microbial community impacts. These do not solely result from OWTS effluent. However, OWTS can be a source of nutrients and therefore can contribute to these impacts.

Nutrients and algae growth

Nutrients can facilitate large, monospecific macroalgae and turf algae blooms (Lapointe, 1997; River and Edmunds, 2001; Smith et al., 2006) that can compete for space and light. The increased macroalgae can prevent sunlight from entering which is detrimental to the coral (Lapointe, 1997; Smith et al., 2006; Vega Thurber et al., 2014). This can also result in multiple adverse impacts on the coral microbiome, such as the depletion of local oxygen concentration (Haas et al., 2010; Haas et al., 2011; Smith et al., 2006), transferal of exotoxins (Barott et al., 2011; Rasher and Hay, 2010) and transmission of pathogens (Nugues et al., 2004; Thurber et al., 2012). Algal blooms can ultimately lead to coral loss and sometimes the death of entire colonies (Rosenberg et al., 2007; Smith et al., 2006).

Coral disease and bleaching

Several past studies have identified a positive correlation between the prevalence of coral disease and the extent of bleaching on natural reefs due to nutrient availability in ambient water (Haapkylä et al., 2011; Kaczmarsky and Richardson, 2010; Richmond, 2015; Shantz and Burkepile, 2014; Vega Thurber et al., 2014). For example, yellow blotch and black band diseases are associated with elevated nutrient levels in ambient water (Bruno et al., 2003; Sunagawa et al., 2009; Voss and Richardson, 2006). Nutrients can also decrease coral growth by elevating the abundance of autotrophic algal partner *Symbiodinium*, which is an organism living in symbiosis with corals (Wooldridge, 2009).

Another significant impact of anthropogenic nitrate inputs to marine waters is coral bleaching (Sunagawa et al., 2009; Vega Thurber et al., 2014). A positive correlation between bleaching prevalence and inorganic nitrogen levels has been reported in the Great Barrier Reef (Wooldridge and Done, 2009) and the Florida Keys (Wagner et al., 2010). The bleaching results in bright and vibrant coral turning white reducing its aesthetic quality.

Coral growth changes

High nutrient concentration in ambient water can alter animal-algal symbioses, shift competitive interactions, cause mortality reproductive failures, and result in inadequate recruitment of juveniles (Richmond, 2015). Shantz and Burkepile (2014) reported an 11% reduction in calcification rates with increased DIN. On the other hand, phosphate in the water positively impacts (9% increase) calcification (Shantz and Burkepile, 2014). The increased calcification is attributed to the deposition of calcium phosphate with the input of additional phosphate in the water. However, an increase in calcination does not necessarily mean the growth of corals.

Microbial community impacts

The significant impact of nitrate in coastal waters could cause shifts in reef community composition (Hunter and Evans, 1995). Nitrate can reduce the growth and survival of adult corals (Hughes et al., 2007; Kinsey and Davies, 1979; Koop et al., 2001) and prevent the recruitment and establishment of juveniles (Hughes et al., 2007; Lirman, 2001; McCook, 2001). This leads to a depauperate state of corals (Richmond, 2015; Sunagawa et al., 2009) and reduces the aesthetic quality of corals. The depauperate state of corals can also impact the main habitat structure of corals.

Appendix B: Sewage associated waterborne microbial diseases

Table B1 Common diseases caused by human enteric pathogens

Pathogen	Disease & Symptoms	Prevalence	Effective removal methods
Gastrointestinal illness			
Adenoviruses (31 types)	Conjunctivitis- gastroenteritis, Urethritis, Haemorrhagic cystitis, Epidemic keratoconjunctivitis Pharyngoconjunctival fever	Worldwide. Not in Australia	Chlorination, Media filtration (with coagulation) Membrane filtration (90%)
Noroviruses	Gastroenteritis- diarrhoea, vomiting, abdominal pain, cramping, low fever, headache, nausea, tiredness (malaise), muscle pain (myalgia)	Worldwide	Filtration Chlorine and UV
Enteroviruses (67 types)	Gastroenteritis –infantile diarrhoea, Acute haemorrhagic conjunctivitis Aseptic meningitis- herpangina, paralysis, exanthema hand foot and mouth disease common cold	Worldwide. Common in Australia	Chlorine and UV Media filtration (with coagulation) Membrane filtration
Hepatitis virus	Fever, nausea, abdominal pain, anorexia and malaise associated with mild diarrhoea, arthralgias, scleral icterus; cytologic damage, necrosis and inflammation of the liver (HAV)	Hepatitis A -Worldwide. Not common in Australia. Hepatitis E- Not in Australia	Media Filtration Disinfection
Rotaviruses	Severe diarrhoea among young children Vomiting, abdominal distress Dehydration, Fever	Worldwide. Very common in Australia	Chlorine and UV Media filtration Membrane filtration
Campylobacter spp.	Gastroenteritis		The soil treatment system can provide adequate treatment.
Salmonella spp.	Gastroenteritis		The soil treatment system can provide adequate treatment.
Shigella spp.	Bacillary dysentery		The soil treatment system can provide adequate treatment.

Pathogen	Disease & Symptoms	Prevalence	Effective removal methods
Cryptosporidium spp. oocysts	Diarrhoea		The soil treatment system can provide adequate treatment.
Entamoeba histolytica	Amoebic dysentery		The soil treatment system can provide adequate treatment.
Giardia lamblia cysts	Diarrhoea		The soil treatment system can provide adequate treatment.
	Central nervous system infections		
Naegleria fowleri	fulminant, almost invariably fatal, amoebic meningoencephalitis after swimming in warm fresh waters	Found in Australia, associated with recreational water	The soil treatment system can provide adequate treatment.
Acanthamoeba spp.	Fatal granulomatous encephalitis	Worldwide immunosuppressed people recreational water	The soil treatment system can provide adequate treatment.
	Respiratory diseases		
Adenovirus *Leptospira* spp.	Complex lung disease Acute respiratory problems Pneumonia,	Immunosuppressed people	Chlorination, Media filtration (with coagulation) Membrane filtration (90%)
	Other		
Acanthamoeba spp.	Liver or renal disease	skin contact with water contaminated with animal (especially rodent) urine	The soil treatment system provides sufficient treatment
Pseudomonas aeruginosa, *Staphylococcus aureus*	Keratitis	people with corneal abrasions	The soil treatment system can provide adequate treatment.
mycobacteria *Mycobacterium ulcerans*	Ear infections	otitis externa and otitis media	The soil treatment system can provide adequate treatment.

Appendix C: Factors influencing pathogen removal from OWTS

Factors Influencing Pathogen Removal in Soil absorption systems

Human enteric pathogens in wastewater include bacteria, viruses, protozoa, and helminths (USEPA, 2002). Water contaminated with wastewater-borne enteric pathogens can result in a range of human and ecosystem health impacts. Hepatitis A, *giardia, shigella*, pathogenic *Escherichia coli*, and *Salmonella* are some of the common disease-causing wastewater-borne enteric pathogens (NHMRC 2018). Additionally, recreational use of waters contaminated with wastewater-borne enteric pathogens can result in infections such as skin rashes and eye diseases. Wastewater-borne enteric pathogens have also been identified with coral disease. Detailed human and ecosystem health impacts and related environmental value depletion are discussed in Chapter 2 of this report.

Human faeces are the main source of pathogens in wastewater. The typical pathogen counts in wastewater and septic tank effluent are given in Table C1 below. Greater removal of helminth and protozoa can be achieved with sedimentation (Ferguson et al., 2009; Gerba, 2008; Zhang et al., 2009). The concentration of bacteria and viruses present in wastewater is several times greater than the number required (infectious dose of pathogens) for causing infection. For example, *Salmonellae* concentration in septic tank effluent is 0-10^7/100 mL, whereas its infectious dose is 15 – 20 number of organisms in 100 mL. This confirms that OWTS cannot be relied on to remove pathogens to a level that does not impact human health.

Table C1 Typical pathogen counts in raw domestic wastewater and septic tank effluent (Adapted from Lusk et al., 2017)

Bacteria	No. in wastewater (/100 ml)	No. in effluent (/100 ml)	Infectious dose (No. of organisms)
Bacteria			
Total bacteria	1×10^8		-
Total coliform	2×10^6		-
Fecal coliform	3×10^4		-
Fecal streptococci	3×10^4		
Escherichia coli		$10^5 – 10^8$	10 for O157:H7
Salmonellae		$0 – 10^7$	15 – 20
Shigellae		$0 – 10^7$	10
Vibrio cholerae		$0 – 10^7$	10^6
Protozoa			
Cryptosporidium parvum		$10 – 10^3$	1
Giardia lamblia		$10^3 – 10^4$	1
Entamoeba histolytica		$0 – 10^5$	1
Virus			
Enterovirus	$10^3 – 10^4$	$10^3 – 10^4$	1
Rotavirus	20 – 100		
Helminths			
Roundworm	5 - 100	-	-

Therefore, the role of a soil treatment system in pathogen removal is critical. In the soil treatment system, pathogens, particularly bacteria and viruses are removed by die-off and physical straining or adsorption to soil particle surfaces (Stevik et al., 2004). Blockage of pathogen movement through the soil treatment system is referred to as straining. The movement of relatively larger pathogens, such as protozoan cysts and helminth ova can be inactivated by this mechanism. However, straining does not ensure the die-off of protozoan cysts and helminth ova as they are protected by outer coatings. Adsorption to soil particles limits the mobility of smaller pathogens such as bacteria and viruses, and eventually inactivates them by die-off. The adsorption of pathogens in the soil treatment system and eventual die-off mechanism highly depends on the soil characteristics such as soil temperature, pH, organic matter content and moisture content. Based on the outcomes of past studies, Table C2 summarises the soil characteristics that play an influential role in the inactivation of pathogens in soil treatment systems.

OWTS system density will also influence pathogen removal in the soil treatment system. High OWTS density will release a correspondingly high number of pathogens to the subsurface. This could exceed the capacity of the soil to stain or absorb pathogens. However, there is a limited understanding of the influence of OWTS density on pathogen removal. This constitutes an important knowledge gap which requires further research.

Table C2 Influence of soil characteristics on pathogen removal

Soil characteristic	Impact	References
Temperature	Bacterial die-off varies seasonally and peaks in summer when the temperature is between 14 – 22℃	Motz et al. (2012)
	Lower bacterial survival at higher temperatures	Jamieson et al. (2002); Sen (2011)
	Protozoa inactivation is higher at high soil temperatures	Peng et al. (2008)
	Virus inactivation is higher at 20–35℃ compared to 4–20℃	Davies et al.(2006)
	The effect of temperature on viral inactivation may be compounded by other factors, such as moisture content, soil texture, and pH	Davies et al. (2006)
	Optimum enteric bacteria survival is close to 5℃, and significant bacterial die-off starts at 25℃	Stevik et al. (2004)
	Viruses survive longer at a lower temperature	Azadpour-Keeley et al. (2003)
	Bacterial die-off doubles with every 10℃ increase in soil temperature	Reddy et al. (1981)
Soil pH	Enteric bacteria - shorter survival under acidic conditions	Reddy et al. (1981); Jamieson et al. (2002);
	Enteric bacteria and viruses - survive best when soil pH is between 6 – 7	Sobsey (1983)
Organic matter content	High die-off rates when organic matter content is low due to high competition for organic matter and high vulnerability to predation	Horswell et al. (2010)
	High organic matter content reduces virus inactivation by competing for soil sorption sites	Gerba and Britton (1984)
Moisture content	Higher moisture content supports bacterial survival for long periods	Jamieson et al. (2002); Tate 1978
	Moist soils can increase the survival of viruses up to 180 days	Yeager and O'Brien (1979)
Presence of other organisms	Poliovirus-1 rapidly inactivated under anaerobic conditions	Hurst et al. (1980)
	Poliovirus-1 rapidly inactivated slowly in the presence of other organisms	Sobsey et al. (1986)
Clay content	High clay content increases the survival of viruses by protecting them from predatory biotic effects	Davies et al. (2006)