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Contamination of estuaries from failing septic tank systems: difficulties in scaling up from monitored individual systems to cumulative impact

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Abstract

Aquaculture in many coastal estuaries is threatened by diffuse sources of runoff from different land use activities. The poor performance of septic tank systems (STS), as well as runoff from agriculture, may contribute to the movement of contaminants through ground and surface waters to estuaries resulting in oyster contamination, and following their consumption, impacts to human health. In monitoring individual STS in sensitive locations, it is possible to show that nutrients and faecal contaminants are transported through the subsurface in sandy soils off-site with little attenuation. At the catchment scale however, there are always difficulties in discerning direct linkages between failing STS and water contamination due to processes such as effluent dilution, adsorption, precipitation and vegetative uptake. There is often substantial complexity in detecting and tracing effluent pathways from diffuse sources to water bodies in field studies. While source tracking as well as monitoring using tracers may assist in identifying potential pathways from STS to surface waters and estuaries, there are difficulties in scaling up from monitored individual systems to identify their contribution to the cumulative impact which may be apparent at the catchment scale. The processes which may be obvious through monitoring and dominate at the individual scale may be masked and not readily discernible at the catchment scale due to impacts from other land use activities.

Keywords Septic tank systems · Contamination · Wastewater management · Aquaculture · Groundwater · Water pollution · Catchment

Introduction

In Australia, there are over one million septic tank systems (STS) with the greatest number in the most populous state, New South Wales (NSW), where there are in excess of 300,000 systems. The most common type of STS in NSW (> 80%) is the septic tank-soil absorption system (ST-SAS) (Beal et al. 2005) with a smaller number (approximately 12%) of aerated treatment systems (ATS). ST-SAS utilise a septic tank and a subsurface trench or bed and rely primarily on the underlying soil for renovation of the primary treated effluent from the tank. ATS are small self-contained proprietary biological treatment systems that rely on mechanical devices to provide mixing, aeration and pumping of the secondary treated effluent which can be land applied by irrigation. Many of the STS in NSW are located along the coastline often adjacent to estuaries where waters may be used for recreation and aquaculture is often practised. In several of these locations, shallow coastal sand beds provide easily accessible water which is used for potable domestic supply.

Surveys and audits of STS and their performance often demonstrate that a substantial proportion perform poorly and may fail over time. The reasons cited include their poor construction, system undersizing with respect to hydraulic loads, or inadequate consideration of soil and land capability assessment criteria in their design (AS/NZS 1547: 2012). Where failures occur, there is concern with respect to their impacts on public health and the environment, although their impacts can be

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uncertain and difficult to quantify. As a result, there have been very few studies which have been able to demonstrate direct linkages between these STS failures, adverse impacts to human health and the quality of receiving waters at the catchment or watershed scale. The inability to discern such linkages and widespread contamination which may be anticipated is due to the fact that not all STS fail all the time and contaminants may attenuate and/or die-off with longer travel times away from systems which are failing. The substantial dilution of effluent after heavy rainfall also results in difficulties in differentiating contaminant pathways from STS in the field. In a study in coastal FL in the USA (where approximately 25% of households use STS with the majority using ST-SAS), Meeroff et al. (2007) reported that they were able to identify the pollutant contribution of STS in coastal canals during seasonally high water table events, but that the impacts to the wider marine environment were uncertain. In a later study, Meeroff et al. (2014) reported that STS impacts on water quality may be measurable in rural coastal areas by comparing water quality in sewered and non-sewered areas, while Sowah et al. (2014) in Georgia studied water quality in 24 catchments and noted that STS can have a significant impact on the pollution of surface waters, particularly during baseflow conditions. In asking the question as to whether STS pose a hidden threat to water quality, Withers et al. (2014) demonstrated that STS may be far more important as chronic pollution sources than their small contribution to stream nutrient loading would suggest. They stated that they can be a major, and potentially underestimated source of water pollution, and that they need to be better regulated and monitored so that they do not threaten water quality in environmentally sensitive areas.

The objective of SAS is to generate unsaturated soil conditions beneath the trench or bed thereby allowing the oxidation of N to NO_3-N, BOD to carbon dioxide and the destruction of most of the faecal bacteria and viral pathogens (Gardner et al. 2006). Provided there is 60 to 90 cm of unsaturated soil before reaching the groundwater table, there is convincing evidence that dissolved organic carbon is reduced by at least an order of magnitude and faecal bacteria/viruses concentrations are reduced by 3 to 4 log from the septic tank. Once the effluent leachate reaches the groundwater, there is further natural disinfection which is influenced by water temperature, travel time and, in some cases, strong viral adsorption on aquifer material (Pang et al. 2003).

During the transit of effluent from the SAS, most of the phosphorus (P) is transformed into the reactive orthophosphate form PO_4-P (Gardner et al. 2006) which is then subsequently immobilised in many soils (Gerritse et al. 1995a), particularly those of fine texture. This capacity for adsorption varies widely from low levels in many sandy soils to high levels in strongly weathered clay soils. If the P adsorption capacity of soils is reached and soils become saturated, P may be exported off-site and into surface or groundwaters. The environmental consequences of the leaching of P tend to be site specific, depending on the depth to the water table, the beneficial use of the aquifer and the separation distance of the SAS to a surface stream. In general terms, P contamination of groundwater from a SAS can occur where the water table is near the surface, the soil is coarse-textured, the hydraulic loading rates are high, the site soil has a low P adsorption capacity, or when this capacity of the soil has been met. If P enters the groundwater it can migrate off-site as has been shown with sewage plumes in groundwater where there are shallow unconsolidated coastal aquifers.

Typically then, nitrate is the most likely chemical contaminant of groundwater underlying unsewered areas, especially those dominated by ST-SAS. It is highly mobile because of its solubility and the rate at which it percolates downward to the groundwater therefore depends on the soil hydraulic conductivity, the depth to the saturated zone and the volume of recharge. Soils with coarser textures such as sands, strong structure, or those under saturated conditions, are the most susceptible to rapid transport of nitrate. Once in the groundwater system, the movement of nitrate is then highly dependent on the hydraulics and hydrogeology of the aquifer; however, it remains in the system unless it is removed by plant uptake or transformed by microbial processes through denitrification.

Nitrate contamination which is also associated with agricultural activities is usually considered more of a threat to shallow aquifers (where they are used for potable supplies) where there are significant numbers of STS, particularly in coastal locations. Septic plumes containing high concentrations of nitrate in shallow groundwater have been found for individually monitored SAS in sandy soils (Robertson et al. 1991; Gerritse et al. 1995b; Geary 2005) with high rates of nitrification recorded usually within metres of SAS. The pathways for contaminant movement from the unsaturated zone below the SAS to the water table are typically preferential with contaminants usually mobilised by rainfall (Nasri et al. 2015). Evidence of nitrate export from STS was observed in a study at the catchment scale by Valiela et al. (1997). Using a mass balance approach where the potential N load to a sandy aquifer in Cape Cod was calculated, the NO_3-N transport was then discounted by loss percentages following an extensive review of the septic plume literature. They showed that 6% of N inputs were lost in the septic tank, a further 35% of the N leaving the septic tank was lost in the leaching field, and another 35% of the N leaving the leaching field was lost within the plume of the septic effluent (i.e. aquifer denitrification). In upscaling this model to a groundwater dominated catchment, they estimated that STS contributed nearly 50% of the average receiving water N load to receiving waters.

Overall though the evidence for increased NO_3-N concentration with increased density of STS is far from convincing. In a review of Australian studies, Whitehead and Geary (2000) noted that there were elevated levels of NO_3-N in groundwater associated with unsewered development in one study, while in other studies in different catchments where STS densities were higher, there appeared to be no clear association between NO_3-N levels (and FC numbers) in groundwaters and the numbers of STS. Local authorities have in some jurisdictions implemented planning controls which limit the density of STS in new developments as providing appropriately sized (and reserve) land area for each SAS can reduce pollution, including nitrate concentrations. There is however disagreement as to what is a sustainable density of STS as this can vary depending on the specific conditions of the land capability constraints where unsewered development is planned. Even studies which have examined this issue, for example Morrissey et al. (2015), did not find any statistical difference in groundwater nitrate concentrations in multi-horizon boreholes drilled upstream and downstream from high density clusters of unsewered housing across a range of different hydrogeological settings, even in the high and extreme vulnerability sites studied.

Apart from the issue of cumulative impact arising primarily from elevated nitrate levels in groundwaters, there is also the potential for transport of the surviving human faecal bacteria and viruses to receiving waters when STS fail or perform poorly. There are concerns in relation to impacts to public health where groundwaters are used for potable supplies and/or where surface waters are used for recreation. While the retention and die-off of most observed pathogenic bacterial indicators occurs within 60 to 90 cm of the infiltrative surface, under certain conditions some bacteria have been found to survive in coarse-grained soils and once in the groundwater, have been observed to survive for considerable lengths of time. Robertson and Edberg (1997) cited a maximum observed transport distance of bacteria in groundwater of 600 m in a sandy aquifer. One recent US study by Schneeberger et al. (2015) reported that where STS discharge effluent into the subsurface, they can be a source of contamination and can impact the microbial quality of shallow groundwaters in coastal locations.

The difficulty at the catchment scale is always being able to determine the cumulative impact that STS have relative to other land use activities in an urban or primarily agricultural area where there may already be impacts to either surface or groundwater quality. As Gardner et al. (2006) suggest, the evidence for off-site impacts from STS in a catchment is anything but clear cut, but there is increasing evidence that this is occurring as recent studies have shown. Difficulties remain however in scaling up from individual systems where localised contamination can be found to catchments where there are significant densities of STS as there are masking effects from nutrient and faecal coliform export from other land use activities.

Research reported in this paper has been undertaken within the community of Salt Ash where there are over 300 dwellings with STS located adjacent to a major shallow aquifer (Tomago Sandbeds) which is used for potable water supply in the nearby city of Newcastle, NSW. The small unsewered community of Salt Ash is mid-way between Williamtown and Bob's Farm adjacent to the Tilligerry Creek estuary (Fig. 1). Performance audits of STS in the area indicated that a significant number had some problems and that several were considered to be in high risk environmentally sensitive areas to both oyster production in the estuary and potable water supply from the aquifer.

The results from monitoring the shallow groundwater quality at one dwelling with a ST-SAS and at an unsewered rural/residential subdivision of 40 dwellings at Salt Ash are reported. In each case, household water use was monitored, and piezometers, networks of suction lysimeters and shallow multi-level bores installed to monitor the migration and transformations of various domestic wastewater contaminants, including nitrogen and phosphorus (and faecal indicators) in the vadose zone and groundwater. In the case of the subdivision, faecal biomarkers such as sterol compounds and their breakdown products were also used to determine if the diffuse sources of contamination from the STS were impacting the shallow groundwater quality and the estuary where there is commercial aquaculture. This technique can be used to distinguish and estimate contributions from various sources of faecal contamination in waters, and sediments and distinguishable sterol profiles for humans, herbivores and birds have been found to be sufficiently distinctive to be of diagnostic value in determining whether faecal pollution is of human or animal origin (Leeming et al. 1996). All faecal material contains sterols and their breakdown products, stanols. The distribution of sterols found in faeces, and hence their source-specificity, is caused by a combination of diet, an animal's ability to synthesise its own sterols, and the conversion of sterols by intestinal microbiota in the digestive tract. Apart from the acknowledged potential to contaminate the shallow aquifer nearby, the principal threat to oyster production therefore appears to be the contamination from failing STS. The issue of cumulative impact and scale is discussed following a review of the results contained in the paper.

Tomago Sandbeds and Tilligerry Creek

The Tomago Sandbeds were formed during the Pleistocene era as an inner barrier dune system with the original sand deposits occurring up to 250,000 years ago. They provide an underground water source which contributes about 20% of the potable water supplies to Newcastle and urban communities in the lower Hunter Valley. The sand beds are parallel to the coast between Newcastle and Port Stephens, starting at Tomago and extending north-east for 32 km to near Nelson Bay (Fig.1). They are between 4 and 14 km wide and approximately

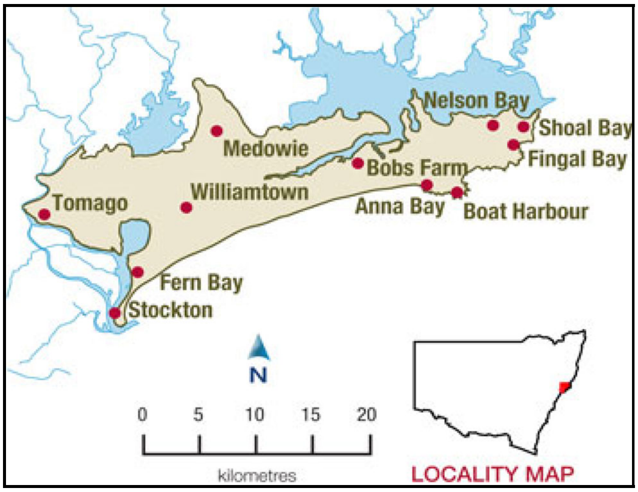

Fig. 1 Tomago Sandbeds (NSW, Australia)

150 km^2 in area. Hydraulic conductivity values for the Tomago Sandbeds have been reported as between 10 to 20 m/day. An extensive system of underground bores draws raw water from the sand beds and pumps it to a water treatment plant prior to reticulation through the Lower Hunter water supply system. The maximum water storage in the aquifer is about 100,000 ML above sea level and it covers an area of about 100 km^2. The water table is approximately 4.8 m above sea level when the sand beds are full and 1.8 m above sea level when empty. The Tomago Sand Member consists of an exposed layer of highly permeable accumulated beach, dune and near-shore shelly sands underlain by the impervious Medowie Clay Member. The thickness of the primarily fine grained sand layer reaches a maximum of 50 m, but on average is 20 m deep. The source of the water in the aquifer is rainfall that lands directly on the sand surface (annual rainfall averages 1125 mm), and while a proportion of the rainfall is lost to plants and evaporation, sufficient water is stored in the sand to provide a viable and significant source of water for ongoing extraction. There are a number of groundwater-dependent ecosystems in the area, including terrestrial vegetation and wetlands, and to protect the high quality of the Tomago Sandbeds drinking water source, public access to the land within the catchment is restricted.

While a significant part of the sand bed area is closed to ensure protection of extraction infrastructure and to maintain groundwater quality, there is some agriculture and residential development adjacent to the Tomago Sandbeds. These land use activities include grazing animals, an intensive poultry farm, a significant number of unsewered premises in the communities of Salt Ash and Bob's Farm, and an oyster growing area within Tilligerry Creek. The narrow estuary is ringed by mangroves and is poorly flushed by drainage waters from the creek. Generalised water table contours on the Tomago Sandbeds in the vicinity of Salt Ash show groundwater levels between 0 and 3 m below the surface with interpreted flow direction from the Sandbeds towards Tilligerry Creek. The Tilligerry area is typically low-lying estuarine country overlying the older Pleistocene Tomago Sandbeds (below 1 m above sea level) and dissected by open drainage channels, some of which are influenced by tidal movements. As a large proportion of rainfall runs off from the catchment, an extensive network of agricultural drains has been constructed to take drainage waters to the estuary. Some of these drains have floodgates to protect against tidal ingress and some run through the unsewered community of Salt Ash to the estuary. Salt Ash, Tilligerry Creek and a number of the major drains entering the estuary near the field sites are shown in Fig. 2. Runoff quality from the combination of catchment land uses which enters the estuary, particularly following heavy rainfall, is typically poor.

Tilligerry Creek is part of the larger Port Stephens estuary and the second largest producer of oysters in NSW with an annual value in excess of A$5 million. Areas within the Tilligerry estuary have on occasions previously been closed to commercial oyster harvesting, particularly following heavy rainfall when runoff containing faecal contamination enters the waterway (Geary and Davies 2003). In 2005, there was a highly publicised case of contaminated oysters as samples of oyster tissue tested positive for human virus in Zone 5A (Fig. 2). Although in this instance there were no impacts to public health associated with their consumption, parts of the estuary remained closed to commercial harvesting for 2 years resulting in substantial loss of income to the primary producers. Since this incident Zone 5A of the estuary has been permanently closed to commercial oyster harvesting by the NSW Food Authority (Shellfish Quality Assurance Program) due to the ongoing issues regarding faecal contamination in these waters and the oyster tissues. While agricultural sources of contamination can be significant in terms of the overall faecal load to this estuary, the contribution associated with failing STS is of concern given that the oyster contamination recorded was from human faecal waste (Kardamanidis et al. 2009; Geary et al. 2015), and this has affected the viability of aquaculture in this part of the estuary overall.

Materials and methods

Field site G

Detailed investigations involving monitoring the movement and fate of contaminants from a ST-SAS were undertaken at one location in the Salt Ash area as shown in Fig. 3 (site G). Results from monitoring at site G, which is located on the margins of the Tomago Sandbeds and considered a high risk area because of its proximity to Tilligerry Creek estuary, are reported in this paper.

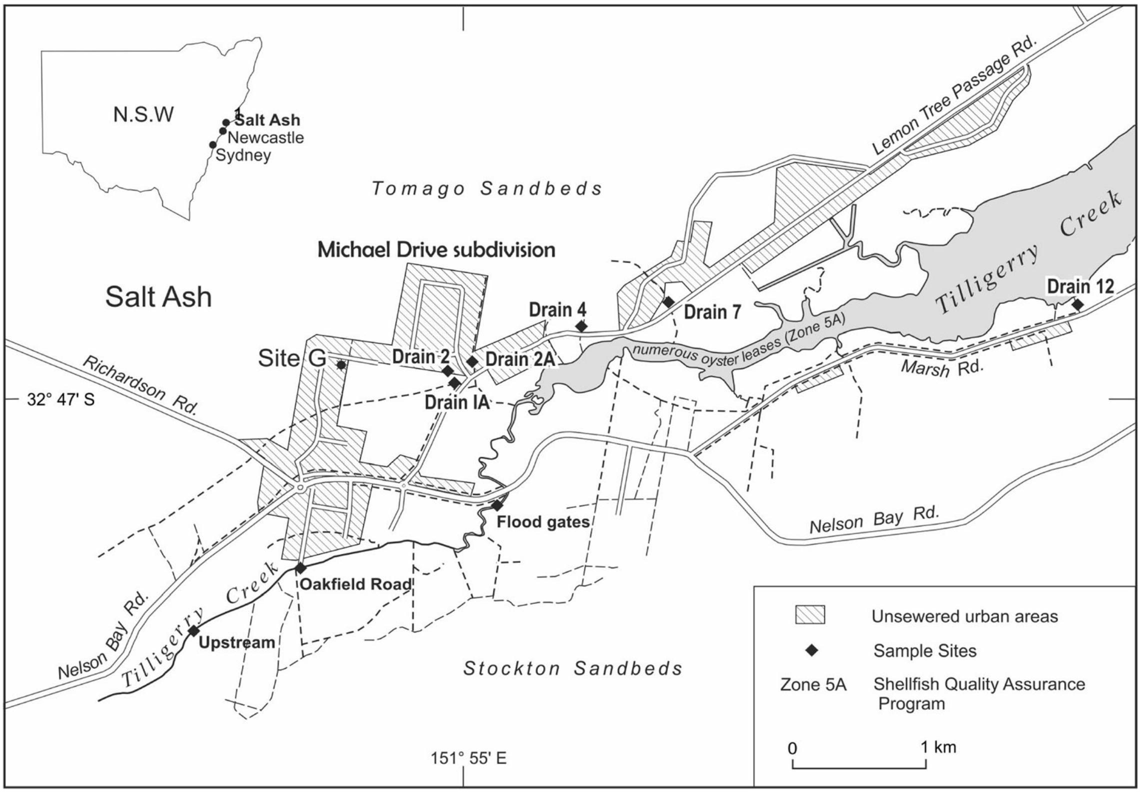

Fig. 2 Salt Ash and Tilligerry Creek showing unsewered areas with STS, sampling site locations and oyster lease areas

The ST-SAS at this site consisted of a 2300 L septic tank with 9 m tunnel trench located approximately 9–12 m from an open creek which downstream entered drain 1A. Suction lysimeters and piezometers were installed downgradient from the SAS at this site as shown in Fig. 3. The suction lysimeters of two lengths (either 70 or 100 cm long) were chosen for use because sands have high hydraulic conductivities, and at low suctions, it was expected that sufficient sample volumes would be collected for chemical analysis. The single chamber vacuum-operated ceramic cup samplers (Soil Moisture Equipment Corporation Model 1900 L) were installed in both the vadose zone and below the water table both inside and outside the property boundary fence line and adjacent to a nearby creek shown in Fig. 3.

Soil water and groundwater samples along with septic tank effluent samples were regularly collected during the 6-month monitoring period (particularly after rainfall) and the detailed chemistry of the effluent determined. The samplers, however, are not considered suitable for faecal indicators because of the difficulty in sustaining sterile conditions (Kresjl et al. 1994), and bacterial samples were only collected from the septic tank and the surface water in the adjacent creek. Work had previously been undertaken to initially determine the direction of subsurface flow in the vicinity of the SAS and measure the hydraulic gradient at the site. The distribution and boundaries of the effluent plume in the vicinity of the drain and riparian zone were also identified by tracer additions as well as sampling. Two piezometers (P1 and P2) made from slotted 20 cm diameter PVC pipe were also inserted into the groundwater and depth to groundwater was regularly monitored at each piezometer using an electronic level measure.

A scaled cross-section of the field site along the approximate A-A' section line shown in Fig. 3 is shown in Fig. 4. The variation in groundwater levels during monitoring, as well as the position of the samplers along this section are also shown. Depending on the groundwater levels at the time of sampling, water samples for analysis were collected either from the capillary fringe or within the shallow groundwater table, or in the case of sampler L5 (shown in Fig. 4), they were not able to be collected.

The 3–4 bedroom dwelling was located on approximately 1 ha and was occupied by two older adults during the study. Household fixtures included a dishwasher, two toilets and two showers, a rainwater tank was used for all potable use and a groundwater bore was available for garden watering. Household water use was measured using a water meter installed on the property and rainfall was manually recorded at the field site during the study period. Using the elevated effluent concentrations of electrical conductivity and nutrients, it was possible to delineate the lateral extent of the STS impacted groundwater at this field site.

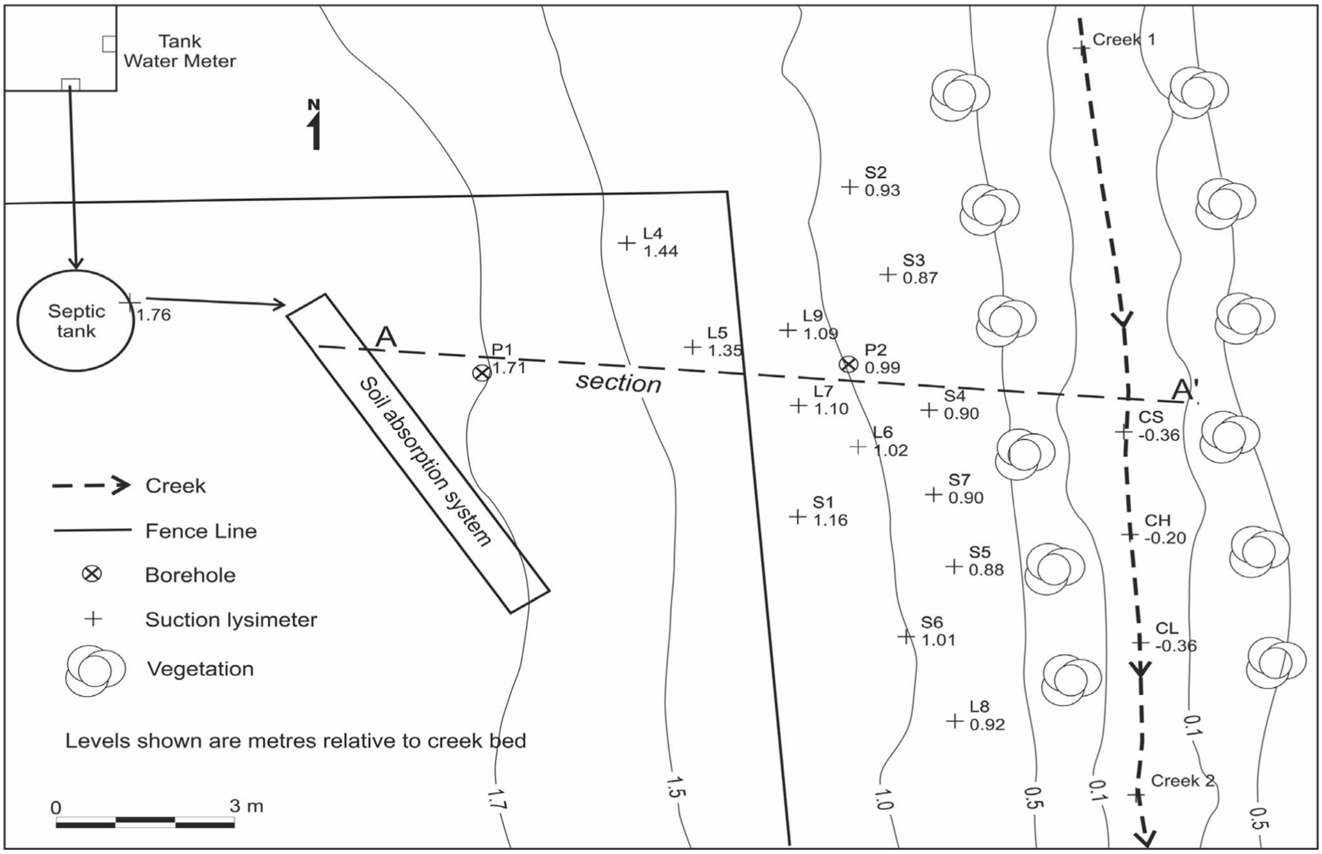

Fig. 3 Field instrumentation site G

Field site at Michael Drive subdivision

This field site was located north of Salt Ash adjacent to part of the Tilligerry Creek estuary shown as the rectangular unsewered area between site G and drain 4 on Fig. 2. Within the Michael Drive subdivision, there are 40 one hectare properties (Fig. 5) which currently use large rainwater tanks for all indoor uses and groundwater extraction for outdoor uses. The STS used by the majority of dwellings are ST-SAS, while several homes have secondary treatment systems; either an ATS or an above ground mound system. So, the STS density for the study area is approximately one wastewater

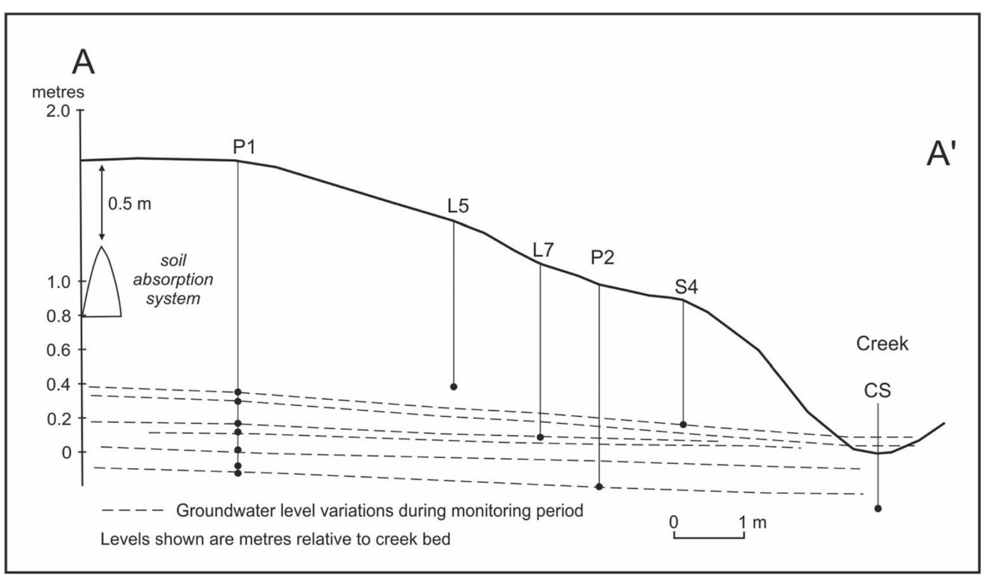

Fig. 4 Topographic cross-section along approximate line A-A' showing position of intersected suction lysimeters and piezometers (levels relative to creek bed) and groundwater level variation during monitoring (VE 2.5)

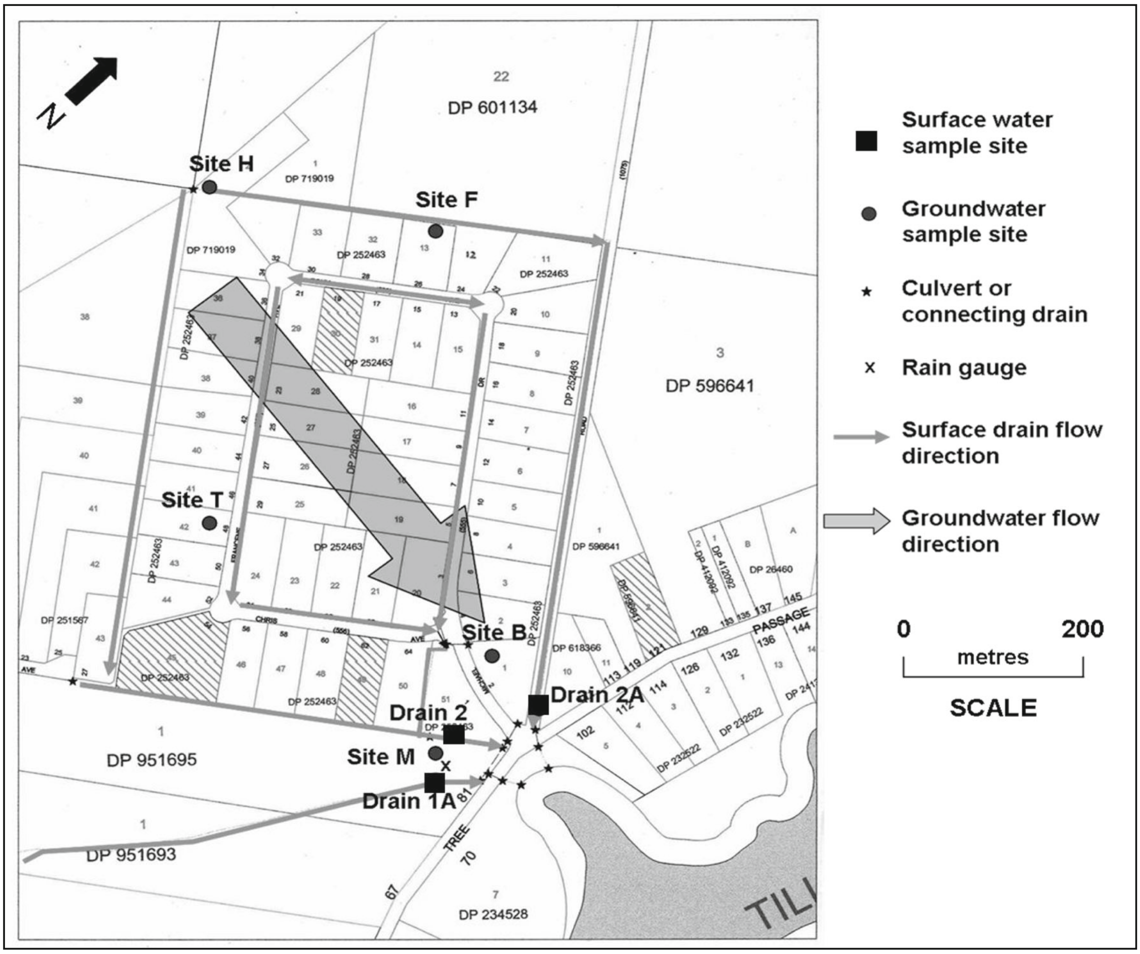

Fig. 5 Field site instrumentation at Michael Drive subdivision

system per hectare. This field site comprised the sandy soils of the Tomago Sandbeds but also included estuarine muds near the estuary margin. The monitoring work was undertaken to determine if the contaminants from the STS in the subdivision were able to be detected in the surface and groundwaters and whether these hydrological pathways could be contributing to contamination within the estuary. Surface water monitoring was undertaken at a number of sites including sites 1A, 2 and 2A and groundwater sampled at five locations (Fig. 5). The groundwater monitoring results are only presented in this paper.

The groundwater bores were drilled and a multi-depth sampling configuration used (0.7, 1.3 and 1.9 m below ground surface) which allowed for three samples to be collected from each depth at each of the five sites. Water samples were collected at sites M, B, F, H and T over a 6-month period, although variable groundwater levels meant that some samples at depths closest to the surface were not always able to be collected at each site. Figure 5 shows the surface drainage and monitoring points in the study area as well as the direction of groundwater movement, culverts and connecting drains and existing drainage lines within the subdivision (Lucas et al. 2007). The groundwater monitoring sites were not located next to or near each STS, but were located at sites selected to represent groundwater leaving the catchment in the direction of the estuary. Two sites (F, H) were located hydraulically above the subdivision and two were located downgradient and along the direction of flow within the subdivision (T, B), while Site M was downgradient and outside the subdivision near the margin of the estuary. Rainfall was also continuously monitored using a 0.2 mm tipping bucket rain gauge which was located near site M, and household water use was monitored using 'smartmeters' at the rainwater tanks at sites M, B, F and T. Water use could not be monitored at the residence at site H. Groundwater level monitoring using a pressure transducer was undertaken throughout the 6-month study period at site M and water samples collected at each site (sites M, B, F, T and H). The water quality analyses undertaken were for a range of chemical and microbiological parameters, as well as faecal biomarkers such as sterol compounds, to determine if there were human-sourced contaminants in the surface and groundwaters at the sites. A suite of sterol compounds was analysed including coprostanol, which constitutes about 60% of the total sterols in human faeces, and the ratios of coprostanol to other faecal sterols calculated to attribute percentage contributions from detected sources.

Results

Field site G

The results of water use monitoring at site G indicated that the residents' average water use was 301 L/day during the 6-month study period with an effluent loading to the STS of approximately 55 L/m^2/day (55 mm/day). The small soil absorption system (9 m tunnel trench) was clearly undersized and the loading rate excessive for such a highly permeable soil. At the time, there was concern at the ability of this sandy soil to adequately treat the effluent on-site prior to its entry into the shallow groundwater table, and the presence of a nearby creek approximately 9–12 m away from the SAS. Very little rainfall was recorded during the monitoring period and, as a consequence, groundwater levels dropped rapidly (Fig. 4) and the adjacent creek ceased to receive upstream surface flow. The sole contribution of flow to the creek for part of the study was the subsurface movement of effluent from the SAS. Tracer testing which was undertaken during the study has previously been reported by Geary (2005), and this work showed a clear hydraulic connection between the SAS, the suction lysimeters and the adjacent creek. Using the peak-to-peak travel times of the tracer, the velocity was estimated at 0.4 m/day and the saturated hydraulic conductivity of the soil was calculated as 14 m/day, which is within the range recorded for the Tomago Sandbeds.

Effluent quality from the septic tank was typical of primary treated domestic wastewater (Table 1). The pH was neutral and the electrical conductivity was also relatively consistent around 1400–1500 μS/cm. Almost all of the nitrogen in the tank was present as ammonium and organic nitrogen (total Kjeldahl nitrogen) with little free oxygen being available. As a result, nitrite and nitrate nitrogen concentrations in the tank were negligible. Total nitrogen results (TKN + nitrate nitrogen + nitrite nitrogen) averaged 140 mg/L in the septic tank. The total phosphorus results in the effluent were high (arithmetic mean 18.4 mg/L) with most of the phosphorus being present as the inorganic orthophosphate.

Using the results of the monitoring data, it was possible to delineate the lateral extent of the impacted groundwater associated with the use of the SAS at site G. Samplers L7, L6 and S4 were located along the approximate plume flow core path while samplers S7 and S5 were downgradient. The mean concentrations for these sampling locations near the plume centerline, as well as sampler CS in the creek bed are shown in Table 1. In addition to pH and EC, analytical results are shown for nitrogen (ammonium and nitrate) and orthophosphate in waters sampled. While nitrite analyses were undertaken, the concentrations were typically low, and although they have not been included in this data table, they are part of the total inorganic nitrogen (TIN) concentrations shown (NO_2–N + NO_3-N +NH_4 –N).

After effluent is discharged through the infiltrative surface of an SAS, the movement of the chemical constituents is dependent on the hydraulics of the groundwater; however, they remain in the system unless removed or transformed by processes such as dilution, adsorption, precipitation, vegetative uptake or microbial activity. At this field site, the highly oxidised septic system plume was approximately 10 m wide downgradient from the SAS. The plume boundaries were well-defined extending laterally between, but not including, samplers S2 and L8. As sampler S2 was hydraulically upgradient from the SAS, the results which have been included in Table 1 are considered background. With respect to samplers L7, L6, S7 and S4, which were only several meters from the SAS and along the path of subsurface effluent movement, aerobic conditions in the sandy soil resulted in high concentrations of nitrate and lowered concentrations of ammonium. The evidence for rapid nitrification is also supported by the increase in acidity as pH is depressed due to the oxidation of organic carbon and ammonium.

Total inorganic nitrogen (TIN) did not substantially decrease downgradient from the SAS system suggesting that inorganic nitrogen was not lost or attenuated through the sandy soil. In contrast however, there was a significant reduction in nitrate nitrogen between samplers through the treed riparian zone to the nearby creek (Fig. 3). At this site, there were still elevated levels of ammonium, but nitrate nitrogen was substantially reduced suggesting that the riparian zone consisting of a line of large paperbark trees (*Melaleuca quinquernervia*) was assisting with nitrate removal. Within the adjacent creek, where sampler CS was only meters from sampler S4, nitrate concentrations were substantially reduced from a mean of 75.2 mg/L at S4 to 2.83 mg/L at CS, while ammonium, which was still present in relatively high concentrations, was reduced from a mean of 32.1 mg/L (S4) to 12.2 mg/L (CS). Although fewer groundwater samples were collected at sampler CS in this part of the study, TIN concentrations appeared to be significantly lower at this sampler relative to sampler S4. Other removal processes apart from plant uptake and dilution could include denitrification which was reported by Beal et al. (2005) to occur in small anaerobic pockets in the vadose zone; however, this was not investigated further in this study. While there is very little available organic carbon for use as an energy source in groundwater, this is not the case in the vadose zone, or in the root zone of the large paperbark trees as their roots are routinely known to exude carbonaceous materials.

Table 1 Septic tank (ST) and groundwater quality at selected sample locations—site G

Sample site		pH	EC (μS/cm)	PO_4-P (mg/L)	NH_4-N (mg/L)	NO_3-N (mg/L)	TIN (mg/L)	FC (cfu/100 mL)	FS* (cfu/100 mL)
Septic tank	Mean	7.44	1480	15.0	108.0	0.20	108.2	1.3×10^5	1.3×10^5
	SD	0.23	131	1.76	15.7	0.15	15.7	2.2×10^5	2.2×10^5
	N	17	17	16	14	10	14	4	4
S2(Background)	Mean	5.40	199	ND	0.79	0.07	0.86	NA	NA
	SD	0.73	82		0.34	0.36	0.36		
	N	3	3	3	3	3	3		
L7	Mean	5.20	968	16.1	30.6	63.1	96.2	NA	NA
	SD	0.82	255	3.79	15.2	18.8	24.2		
	N	16	16	16	15	16	15		
L6	Mean	5.74	1119	12.9	34.4	64.4	103.9	NA	NA
	SD	0.96	251	4.6	16.3	33.4	28.4		
	N	16	16	15	13	15	13		
S4	Mean	4.79	1130	17.2	32.1	75.2	108.7	NA	NA
	SD	1.19	178	2.62	21.7	25.1	17.9		
	N	25	25	25	22	25	22		
S7	Mean	4.89	1063	16.6	37.6	64.7	102.4	NA	NA
	SD	1.10	233	2.38	16.9	20.6	26.9		
	N	14	14	14	14	14	14		
S5	Mean	4.94	874	14.4	30.0	46.3	77.4	NA	NA
	SD	1.17	193	4.36	20.1	12.0	25.8		
	N	22	22	22	21	22	21		
CS	Mean	4.85	432	3.29	12.2	2.83	15.1	6.1×10^2	9.0×10^2
	SD	0.09	93	0.61	1.36	1.16	2.0	4.5×10^2	
	N	8	8	8	4	4	4	3	2

*Mean, *SD* standard deviation, *N* number of samples for pH; *EC* electrical conductivity; *PO_4-P* orthophoshate, *NO_3–N* nitrate nitrogen, *NH_4 –N* ammonium nitrogen, *TIN* total inorganic nitrogen (NO_2–N +NO_3-N +NH_4 –N), *FC* faecal coliforms, *FS* faecal streptococci, *ND* not detected, *NA* not analysed

The fate and transport of phosphorus are controlled by precipitation and sorption reactions in the soil, and where there are sandy soils with low adsorption capability, phosphorus can move rapidly to the groundwater. This is the case at site G where the phosphorus adsorption of the Tomago Sands has been shown to be low (15 mg/kg) and the orthophosphate is readily transported from the SAS system to the groundwater and the adjoining creek. Total phosphorus results in the septic tank were high (arithmetic mean 18.5 mg/L) with most of the P being present as inorganic orthophosphate. Apart from the background site (S2) where orthophosphate was not detected, other sampler results in Table 1 indicate high orthophosphate concentrations downgradient from the SAS. Within the plume core, the mean orthophosphate concentrations for a number of samplers were also high (for example, samplers S4 (17.2 mg/L) and S7 (16.6 mg/L)) indicating little removal or attenuation during transport. There were however lower concentrations in the drain again suggesting that as the effluent moves away from the SAS and through the riparian zone, there is the opportunity for plant uptake of this nutrient as well as dilution.

While the groundwater samples collected during this study were not analysed for faecal coliforms (due to the inability of the samplers to maintain sterile conditions), effluent samples from the septic tank and the surface water in the creek downgradient from the SAS were collected on several occasions for bacteriological analysis. The hydraulic connection had previously been established following tracer work, and because of the dry conditions during the monitoring period, it was confirmed that effluent was moving through the SAS into the creek. During these times the creek water was odorous and the electrical conductivity elevated suggesting that effluent from the SAS was the source of this water. There was approximately a 3-log reduction in faecal coliform and faecal streptococci numbers following subsurface transport of the effluent from the septic tank and through the soil to the surface water in the creek.

Field site at Michael Drive subdivision

During this field site study at the Michael Drive subdivision, the hydrological connection between rainfall, groundwater levels and runoff in surface drains was clearly identified in the monitoring undertaken at site M. Figure 6 shows a 6-week period of monitoring which highlights the consistent cyclic nature of changes in groundwater and surface water levels and suggests that tidal influence also plays a role at this location close to the estuary margin. The rapid response of the groundwater levels and surface drain levels to incident rainfall can also be seen in the figure.

The data from the 'smartmeters' showed the volume and timing of water used at four households and has been used as a surrogate for daily wastewater generated by each STS. The monitoring indicated that the residents' average water use was 216 L/day at site M (occupancy two persons), 309 L/day at site B (occupancy two persons), 413 L/day at site T (occupancy five persons) and 260 L/day at site F (occupancy four persons) during the 6-month study period. These figures of household water use are quite low and reflective of the caution associated with water use when water is sourced from variable rainfall in a non-reticulated subdivision. Each of the households had a ST-SAS with trench lengths of between 9 and 15 m. Average hydraulic loading to the STS in the subdivision based on water used was approximately 34 L/m^2/day (34 mm/day) which, while significantly lower than the single residence monitored in detail at site G, is still relatively high compared to the recommended design loading rate for primary treated effluent in this soil category (AS/NZS 1547: 2012).

Since 40 residences exist in the subdivision and water use was monitored at four, the discharge from STS has been assumed to represent approximately 10% of the total wastewater flows from the subdivision. The average occupancy for monitored residences was 3.3 persons per household. Wastewater contributions from the four residences have been extrapolated to the 40 by multiplying wastewater discharge by ten to provide an estimate of total wastewater discharge from STS in the subdivision. Based on actual monitoring data, the average wastewater produced was calculated at 92 L/p/day. For forty residences at an occupancy rate of 3.3, the number of people contributing to the hydraulic load equalled 132 persons within the subdivision. Therefore, the wastewater generated by the subdivision was approximately 12,144 L which was discharged to the groundwater each day. Considering the subdivision area of 400,000 m^2, the depth of wastewater discharged is equivalent to approximately 0.03 mm/day. Over a 12 month period, this equates to only 11 mm/year which when placed in context of an annual average rainfall suggests that the total wastewater discharges from the subdivision potentially represent approximately 1% of runoff flows in an average rainfall year.

Multi-depth sampling occurred at each of the groundwater sites and the results from the chemical and microbial analysis for each groundwater site are summarised in Table 2. The data obtained by analysing 60 samples have been arranged in order of the proximity of each site to the estuary, for example, site M was nearest the estuary, while site H was furthest from the estuary (Fig. 5). The number used at each location represents whether the samples were collected from the shallowest to deepest multi-sample point, for example F1 was closest to the surface while F3 was furthest from the surface. Sites B1 and T1 did not ever have an adequate volume of water to sample because the groundwater table was lower than the sampler, while Site M1 had water present on two occasions only and Site T (which was installed later following the first sampling occasion), was not sampled in the first round.

The pH of all groundwater samples collected in the subdivision was typically around 5–5.5 and electrical conductivity varied between 180 and 1620 μS/cm. In comparison to other sites, EC at sites M and B was approximately three to four times higher than at sites T, F and H, possibly reflecting their

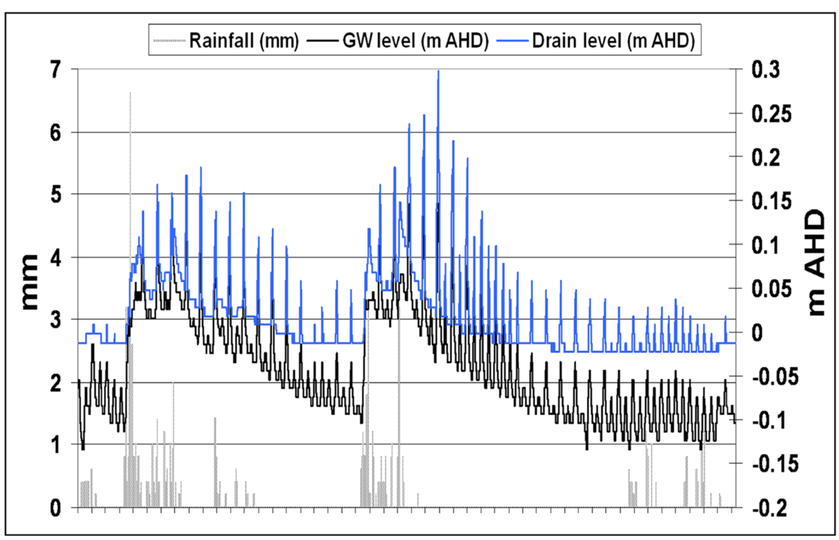

Fig. 6 Monitoring results for rainfall (mm), surface drain level (metres above Australian Height Datum) and groundwater level (m AHD) for a 6-week period at site M—Michael Drive subdivision

Table 2 Groundwater quality at sample sites at Michael Drive subdivision

Sample site		pH	EC (µS/cm)	PO_4-P (mg/L)	NH_4-N (mg/L)	NO_3-N (mg/L)	FC (cfu/100 mL)	FS* (cfu/100 mL)
M1 (*N* = 2)	Mean	4.90	1278	0.06	1.38	0.80	1.0×10^1	NA
M2 (*N* = 5)	Mean	5.03	1055	0.11	0.72	0.54	2.5×10^1	3.5×10^1
	SD	0.73	298	0.07	0.31	0.17	5.0×10^1	2.7×10^1
M3 (*N* = 5)	Mean	5.07	923	0.09	0.54	0.04	1.0×10^0	3.5×10^1
	SD	0.22	251	0.05	0.07	0.17	1.0×10^0	4.0×10^1
B2 (*N* = 5)	Mean	5.71	1431	0.13	1.40	0.70	ND	ND
	SD	0.30	190	0.14	0.59	0.23		
B3 (*N* = 5)	Mean	5.71	837	0.07	0.51	0.54	ND	ND
	SD	0.24	110	0.03	0.12	0.17		
T2 (*N* = 4)	Mean	5.43	473	0.18	0.38	0.53	ND	ND
	SD	0.12	90	0.22	0.28	0.17		
T3 (*N* = 4)	Mean	5.37	295	0.10	0.30	0.60	ND	ND
	SD	0.10	42	0.07	0.02	0.24		
F1 (*N* = 5)	Mean	5.83	360	0.13	0.24	0.38	ND	ND
	SD	0.29	81	0.11	0.18	0.15		
F2 (*N* = 5)	Mean	5.68	203	0.11	0.19	0.50	ND	ND
	SD	0.25	17	0.06	0.06	0.19		
F3 (*N* = 5)	Mean	5.43	188	0.09	0.18	0.38	ND	ND
	SD	0.22	9	0.05	0.04	0.11		
H1 (*N* = 5)	Mean	5.65	263	0.15	0.14	0.64	ND	ND
	SD	0.30	21	0.15	0.03	0.11		
H2 (*N* = 5)	Mean	5.61	262	0.09	0.13	0.68	ND	ND
	SD	0.21	35	0.10	0.07	0.30		
H3 (*N* = 5)	Mean	5.76	270	0.10	0.22	0.74	ND	ND
	SD	0.13	12	0.04	0.13	0.26		

*Mean, *SD* standard deviation, *N* number of samples for pH, *EC* electrical conductivity, *PO_4-P* orthophosphate, *NO_3–N* nitrate nitrogen, *NH_4 –N* ammonium nitrogen *FC* faecal coliforms, *FS* faecal streptococci, *ND* not detected, *NA* not analysed

locations closer to the estuary margin and the ingress of more saline waters. A decrease in EC with depth was observed at all sites. During drilling at several of the sites, a coffee rock layer containing fine indurated sands was encountered, and this had a marked impact on the turbidity and colour of the water sampled. Sites with coffee rock horizons coinciding with groundwater sample depths included M, B and T. Sites H and F were predominantly comprised of sand horizons of varying grain size. At all sites with respect to ammonium, nitrate nitrogen and orthophosphate, groundwater concentrations were low and generally decreased with depth. Nitrogen species concentrations were marginally higher with increasing proximity to the estuary. There were no groundwater sites either above or below the subdivision where water quality concentrations for the nutrients were elevated above background concentrations. While total coliforms existed in appreciable numbers in all the groundwaters sampled, faecal organisms (faecal coliforms and faecal streptococci) were not found in most of the groundwaters sampled, except in several samples collected at site M (Table 2). The absence of these indicator organisms at sites (other than at site M) suggests that land use activities, including the presence of STS, were not contributing these organisms to the groundwater, or that if they were present, they were not able to survive in large numbers.

The faecal sterol analysis results for 68 groundwater samples also resulted in low or negligible concentrations of a number of the important sterol biomarkers. Due to the low concentrations overall and the large number of samples where coprostanol was not detected, the use of ratio analysis to interpret contaminant source was limited. In fact there was only one site where coprostanol was recorded and where ratio analysis could be undertaken. This was at site H2 where a low coprostanol concentration (12 ng/L) was measured. Using the ratio method outlined by Leeming et al. (1996) for the epicoprostanol and coprostanol concentrations measured, the source of contamination in this one sample appeared to be most likely from herbivores rather than humans. Overall, the faecal sterol compound concentrations in groundwaters at this field site were low relative to those levels likely to be found in faecally contaminated waters. On the basis of the low nutrient concentrations in groundwater, the low microbial counts and low or non-detectable coprostanol concentrations in the samples collected at this field site, the groundwater was not considered to be a major pathway for nutrient or microorganism export to the estuary, and the contribution associated with contaminants from STS in the subdivision, was minor overall.

Conclusions

This research has described the monitoring of the subsurface movement of contaminants from STS at two different field sites in a sensitive coastal environment; an individual ST-SAS at one property and a large subdivision consisting of 40 dwellings, each with their own STS. The research highlights the difficulty in scaling up results obtained from monitored individual systems to identifying cumulative impacts at the catchment scale. The contaminant loadings associated with the nature of an effluent discharge from individual STS reflect the change from a point source to multiple diffuse sources as the impacts to groundwater, which are apparent at the individual scale, may be masked at the larger catchment scale. This is primarily because different processes dominate at different scales, and those processes which may be important at a particular scale may be insignificant at another scale (Bierkens et al. 2000).

The results from the field study at site G suggest that it is relatively straightforward to monitor and identify a plume of STS effluent and its subsurface boundaries using chemical analyses of groundwater. At the sensitive coastal location where the research was undertaken, the transport of inorganic nitrogen and orthophosphate from an individual SAS was substantially unattenuated in the highly permeable sandy soils. There was a high rate of nitrification which occurred within metres of the SAS resulting in high nitrate concentrations which moved through the vadose zone and groundwater. The highly mobile nitrate was then partially attenuated or removed when it encountered a riparian zone and discharged into a creek, although other studies in different geological settings have shown limited nitrate removal associated with substantially larger vegetated buffer strips (Robertson and Schiff 2008). While orthophosphate has the ability to be potentially mobile due to its solubility, it is present in lower concentrations and can be attenuated in soil by adsorption and precipitation reactions in the vadose zone. In many coastal locations with sandy soils, the P adsorption capacity of soil is usually very low, so there remains an issue associated with the transport of orthophosphate from STS where it may enter groundwater and also discharge into surface waters. The subsurface water at site G was enriched with P and not substantially attenuated as it was transported away from the SAS to the riparian zone, although lower concentrations were recorded in the adjacent creek, possibly as a result of some vegetative uptake and dilution.

Perhaps of more significance from a public health point of view is the fact that faecal microorganisms from the STS monitored at site G were able to be detected in surface creek water. Work confirmed that this system was responsible for these faecal bacteria in the creek (even with a 3-log reduction in numbers from the septic tank), and the other chemical indicators, such as nitrate, ammonium and orthophosphate, were present at elevated concentrations in the stagnant pools of diluted effluent. With the proximity of the creek to the Tilligerry Creek estuary, there was a clear hydraulic linkage from this STS towards to the creek and potentially the faecal contamination of oysters which is periodically recorded in the estuary.

The research reported from the study at the Michael Drive subdivision was undertaken at a different scale to the work at site G. At this field site, both surface and groundwaters from an unsewered subdivision consisting of 40 STS at a density of 1/ha were sampled and analysed. The data presented for groundwaters in the Michael Drive subdivision indicated that there was little measureable impact associated with human-sourced chemical and microbiological contaminants from STS. Surface flow and not groundwater flow was considered the dominant pathway for contaminant transport from the land to this part of the estuary, but even these data suggest that the surface drains do not contribute significant sources of contaminants from STS to the estuary. Overall though it was not possible to attribute any significant contamination in the estuary to failing STS throughout the monitoring period at this scale, yet individual systems do clearly contribute to the contamination as the research results suggest. It did appear however that most of the faecal material and nutrients being exported to the estuary were related to agricultural activities and the presence of herbivores in the larger Tilligerry catchment, yet human derived pathogens, possibly from failing individual STS have previously been found in the oysters.

There is a clear case for improving the design and sizing of individual STS, so that when systems fail, the contaminants can be attenuated in the subsurface before entering surface and groundwaters. Appropriate land capability assessment needs to be undertaken for each site, and the adoption of adequate vertical separation distances from groundwater and horizontal buffer distances from drainage lines mandated to ensure that contaminants, which can be found at the individual scale, do not contribute to a cumulative impact at the catchment scale. It is very important to maintain riparian vegetation in sandy coastal locations, particularly where there is shallow groundwater and where aquaculture is practised nearby in estuaries. In most local jurisdictions in NSW, there are few regulatory requirements to monitor STS performance after their installation and particularly as the systems age. On-going monitoring of STS should be required and national discharge limits set as recommended by Withers et al. (2014). Notwithstanding the results obtained by Morrissey et al. (2015), the number of new STS needs to be managed through the implementation of land use planning controls to regulate the density of STS. There is still a difficulty however in determining what these threshold STS densities might be given the variable site and soil conditions that exist in catchments everywhere.

At the catchment scale, nutrient exports from individual STS can be masked by other land uses such as agriculture and urban runoff. Upscaling results from individual STS systems to the catchment scale using mass balance approaches appears to result in overestimates of the significance of the nutrient contributions from STS relative to agriculture, however with regard to faecal contamination, one individual STS which is incorrectly sited or failing may alone be responsible for contamination observed at the catchment scale in a sensitive receiving environment. While there is value in undertaking monitoring at both the individual site and catchment scales, there remain difficulties in using the information obtained at one scale to manage at another scale.

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