3. Risk Characterisation

Modelling proved relatively straightforward and appeared to be informative of Hazardous Event risks, and their causes and impacts, arising from limited dilution, plume surfacing, limited inactivation and on-shore transport. The main constraint was very high number of exposure Scenarios which might have been explored.

The proposed tasks summarised in Appendix 05 Final Study Approach, were largely implemented and yield among other things;

1. Contaminant levels to which bathers would be exposed;
2. Estimated risk of illness or infection;

This Chapter reports on the primary risk characterisation outputs and comprises presentation of results on and discussion of:

1. Historical Data Analysis
2. Hydraulic Modelling;
3. Assessment of Risk at Exposure Points under different Exposure Scenarios;
4. Variation in Waste Stream Dilution and Inactivation;
5. The Episodic Character of Contamination Events;
6. Infection and Illness Risk.

Chapter 4 (Key Findings) summarises the principle interpretations made of the risk probability data.

Chapter 5 (Uncertainties) outlines and discusses the kinds of uncertainties we considered as being either most critical to take note of or most of interest to HWC and its stakeholders.

Chapter 6 (Conclusions) draws the information together and present final conclusions arising from the risk assessment.

As with the Chapter 2, the main findings are presented in the main text with additional Appendices providing further detail to explain data analysis and interpretation where needed.

3.1. Historical Data Analysis

3.1.1. Evidence of Hazardous Events/Periods in Routine Indicator Monitoring Data

The first step recommended in the Guidelines where managing bathing risks is to analyse the historic data. Examination of the historical Beachwatch data provided by HWC indicated that the median numbers of enterococci was below assay detection limits (1 enterococci cfu per 100 mL) and the water quality was consistent with Category A (95th percentile enterococci count < 40 /100mL).

Examination of the full historical data set also indicated there were numerous occasions when the enterococci and thermotolerant coliform numbers were markedly elevated at one, or frequently, several beaches concurrently.

Less than half of these periods were associated with rainfall/run-off periods and it was assumed conservatively that these periods were when enterococci in discharge were being transported into the bathing area. To quantify ‘non- rainfall’ v. rainfall derived (and presumably discharged derived)
events further, the 2001-2006 enterococci data set was classified into dry weather and wet weather and event and non-event using the following criteria:

- A ‘rainfall event’ was assumed to occur when there was:
  - >5 mm total rainfall on the day of bacteriological monitoring and the previous day;
  - or
  - 10 mm total on the day of measurement plus the previous 2 days
- A water quality event was defined as a day on which the numbers of enterococci were >90th percentile for that beach for the whole data set including the poorer years prior to 2001.

Rainfall related data were then excised leaving ‘dry weather’ event only. The enterococci counts associated with rainfall events were then graphed on an exceedence probability (Appendix 22: Exceedence Probability Statistics and Risk Benchmarking, Appendix 14: Operational Application of Exceedence Probability Analysis To Hazardous Event Characterization) plot (Figure 3-1).

This data did not have the same capacity for assessing the impact of the low frequency high impact events that modelling did as this would have required many more data measurements. But it did still provide a data set against which model estimates could be compared or calibrated in part. Notable features included:

1. The PDFs from the four reference beaches were virtually identical including Burwood Beach, which is closest to the outfalls supporting the contention that the contaminant source was not localised to one beach.
2. The levels of enterococci exceeded the 100 per 100 mL around the 99.8 percentile (note the scale on the graph is per L for comparison with later plots).

Further details of this analysis are presented in Appendix 01: Newcastle Beachwater Quality and Baseline Reductions.

### 3.1.2. Derivation of ‘Baseline’ Microbial Reduction Estimates

Predicting modal ‘Baseline’ reduction rates for microorganisms in the coastal zone following discharge ideally required a large number of measurements over time of a conservative tracer. The enterococci level reduction were seen as giving an indication of the median barrier effect provided by the ocean outfall disposal systems. It was known that:

1. The enterococci load from the effluent was ca 1 order of magnitude greater than that of the WAS (actual figure determined by resampling was 7X – see Section 2.3);
2. The two waste streams are discharged in the same general area of the coastal zone;
3. Enterococci from secondary effluent and WAS occurred at similar rates from the work on inactivation studies that (Section 2.6.4);
4. From a previous WRL study enterococci appeared to be relatively persistent in the environment ($T_{90}$ of 20 to 50 hours) and was more persistent than $E. coli$ (Glamore et al., 2008) and these rates are comparable to those reported for several viruses in the Guidelines (Table 5.8).

So it was proposed that:

1. The apparent change in enterococci numbers be used as an indication of pathogen reduction under normal favourable hydraulic conditions.
2. Baseline enterococci and pathogen reduction in the swimming zone
   ≈ (Enterococci numbers in secondary effluent + Enterococci numbers of WAS*0.1) / Reduction (factor) provided by the coastal zone.
3. Enterococci numbers reflected a conservative minimum reduction in pathogen numbers.
The initial (log$_{10}$)numbers of enterococci was $ca$ 5.4±0.3 per 100mL in effluent (the dominant source of enterococci). When compared with the geometric mean of enterococci numbers at the beach this indicates a decimal reduction factor of $ca$ 5.12± 0.02 from a comparison with the dry weather effluent quality. Details of work confirming the $=10^5$(effluent) and $=10^6$(WAS) Baseline reductions were reasonable approximations and refinements of this reduction estimate are provided in Appendix 01 Newcastle Beachwater Quality and Baseline Reductions.

It is recognised that some pathogens may be more resilient but it was expected that their potential impact should be indicated by the hazardous event modelling, particularly the Conservative inactivation Scenarios.

![Figure 3-1. Exceedence Probability Plots of Enterococci Numbers for Dry Weather Conditions at the Beaches from Historical Data](image)

Notes:
2. Enterococci counts are normally expressed as cfu per 100mL. For consistency with other data we have expressed all microbiological data in units per L.
3. $n=220$ so limiting percentile which can be estimated 0.995 (99.5%).
4. X axis is shown as 1-Percentile due to crowding of data on standard arithmetic plot.
5. Half of the values were below detection limit, there were treated as half detection limit values for calculation purposes.

3.1.3. Previous Hydraulic Modelling and QMRA

As noted above understanding the coastal zone hydrodynamics and developing the modelling would not have been possible without previous studies (Glamore et al., 2008) (CH2MHill, 2008). The former report has been used extensively. Further assessment of the CH2MHill report was not undertaken as the risk estimates were point ones, and the input data selected were more provisional being taken directly from the literature with only limited opportunity for extensive critiquing. The
latter does not reflect negatively on this assessment in any way as its aim was to determine whether risk levels of concern might occur, as a prelude to a more detailed study. Its conclusion was there probably were concerns and the results of the current study suggest this conclusion.

### 3.2. Hydraulic Modelling Output Statistics

#### 3.2.1. Raw Output Summary Statistics

Examination of the raw hydraulic model outputs indicated it was not uncommon for waste stream contaminant particles (and thus microorganisms) in the swimming zone at a dilution of 1 part in $10^4$ or less during the 3 month timecourse simulated. For example at the 4 beaches considered during the summer months low event dilutions were seen on ca 20 occasions depending on how distinct each event was seen. It also appeared that a more detailed examination of summary statistics prior to the QMRA would allow identification of higher risk scenarios where the QMRA could be focused.

Summary hydraulic modelling statistics for WAS and Secondary Effluent are shown in Appendix 24 Summary of Hydraulic Modeling Outputs and Appendix 25 Summary Tables for Hydraulic Modelling Statistics along with detailed explanations of their significance. In order to interpret theses statistics it is useful to review the attributes of each model data set, the terminology used and how this was used to develop microbial reduction PDFs for each hydraulic modelling scenario.

Each run of the hydraulic model involved the following:

1. Establishment of the scenario to be simulated (no. of potential waste stream/inactivation rate/exposure location/year/season = $2 \times 4 \times 8 \times 2 \times 2 = \text{maximum of 256}$);
2. ‘Injection’ of ‘pathogen’ particles into the model waste streams at increments of 15 minutes over a period of 3 months at a rate of ca $10^{10}$ particles per day. (This yielded ca 8516 fifteen minute timestep estimates of particle dilution+inactivation).
3. For the purposes of the current work each timestep was taken as one random set of water quality conditions a bather would be exposed to;
4. Simulation of the fate of every particle injected covering:
   a. their mixing, dispersion and transport along the Newcastle coastal zone waters in three dimensions towards, along and away from the coastline;
   b. their inactivation due to solar radiation;
   c. ultimately their departure from the Newcastle coast;
5. Collection of aggregate statistics on the particle numbers reaching each model cell corresponding to a beach/exposure points and their remaining ‘mass’ (measure of their inactivation) and their time of travel to the exposure point (also in effect the travel time from the discharge point available for inactivation).
6. Estimating total reduction for the particles detected at each exposure point cell for each 15 minute timestep –
7. Total reduction was subsequently estimated by multiplying:
   a. the dilution factor (number of particles at the exposure point / number of particles in the wastestream); by
   b. the inactivation factor (the extent to which particles observed were estimated to have been reduced in ‘mass’).

The reason for the separation of dilution and inactivation is as follows. Conceptually it would be possible to inject a much larger numbers of particles into the model, and subject the particle population to both dilution and inactivation as they moved from finite element mesh cell to mesh cell. However to cover declined by $>10^6$ orders of magnitude and track them along with all the other
previously inject particles is computationally impractical. So the two steps are separated and each particle is assigned a ‘mass’.

Another computational constraint was that though particles may decrease in mass, they are still present in the model and tend to accumulate over time. For a three month timeseries the accumulating number becomes so large that again their fate and transport and cannot be calculated with current computing technology. As a result it is necessary for older particles to be removed after they have spent a substantial time being tracked within the model corresponding to the point when they would in reality have been inactivated or transported away from the area by large scale coastal currents. Based on past hydraulic modelling experience and first principles WRL removed ‘conservative’ particles after 2 days residence and other particles after 7 days residence. The difference is because numbers of particles in Conservative scenarios tend to accumulate much more rapidly.

The modelling process is outlined further in the accompanying report (Rayner et al., 2009) and in the Methods section above. Illustrative elements of the method are reproduced in Appendix 13 Selected Excerpts from WRL Modelling (Glamore et al., 2008).

### 3.2.2. What Analysis of Hydraulic Data Showed about Risk Reduction by Coastal Waters

The risk related information and insights obtained from the primary hydraulic modelling of the coastal waters ‘barrier’ prior to QMRA, were as follows (see Appendix 24 Summary of Hydraulic Modeling Outputs and Appendix 25 Summary Tables for Hydraulic Modelling Statistics for details):

1. General:
   a. There were only minor differences between the four beaches in the frequency with which particles were detected (and hence when dilution was suboptimal) for scenarios with the same combination of simulation year + season + solar inactivation rate;
   b. There were only minor differences between the particle number and mass ‘detected’ at the 50 and 200 m exposure locations at any given beach;
   c. The major sources of variance in estimated reduction in (pathogen) particle numbers and mass were:
      i. The material type (WAS or effluent);
      ii. The different inactivation rates modelled;
      iii. The summer v. winter transport coastal hydrodynamic inputs;
      iv. Different between day predisposing conditions so that on one day several events might occur, followed by no particle detects for up to a week.
   d. Contamination was episodic with occasionally multiple adjacent timesteps making up a larger event of several hours duration;
   e. The more the radiation exposure, the less frequently particles were detected. Median particle travel times under high radiation conditions tended to be short (< 1 day) because the longer they were in the system the higher the degree of inactivation they experienced. However, complete inactivation was rare and inactivation followed closely the occurrence and intensity of daylight;
   f. The average timestep by timestep travel time of the particles detected ranged from the theoretical maxima (ca 2 and 7 days) to minimum travel times as short as 45 minutes. In a preliminary examination of the data though no strong correlation between travel time and mass and number of particles was detected.
   g. Sometimes more timestep records contained particles under the 75 MJ.m² scenarios than the corresponding conservative scenarios. This was due to particles in the latter
case being ‘programmatically’ removed as noted above. Based on comparison of summary statistics the effect appears minor compared to the other sources of variation.

h. While particle inactivation as high as 4 log_{10} units was noted, the grand average particle inactivation estimates for different scenarios were all <2.

2. Secondary Effluent:
   a. Merewether Baths and Bar Beach tended to be worse than Dudley Beach and Burwood Beach in the reduction achieved in summer effluent particle levels. But the pattern was not so marked in winter.
   b. The highest rate of detection of particles was from modelling of treated effluent which tended to be much more of a concern than for the WAS. Under conservative conditions up to 19.6% of all timesteps of the effluent stream contained some particles and the minimum reduction was only one part in 10^2 compared to the empirically estimated median Baseline of \( ca \ 10^5 \).

3. WAS
   a. In summer, reduction in WAS particle numbers was extensive particularly in the case of the 3 MJ.m\(^{-2}\) scenarios where no detection occurred in some instances;
   b. In winter, reduction was not so marked even with the 3 MJ.m\(^{-2}\) detection of particles occurred in up to 7% of the timesteps.

Provisionally the following were concluded:

1. Diluted effluent reaches the beaches much more frequently than seems to be fully appreciated, \( ca \ 1 \) event per 3 to 4 days in summer.
2. \( In \ extremis \) relatively concentrated material can reach the shoreline.
3. The levels at such times are equivalent to a dilution of less than 10^4 and at such times there is likely an elevated risk of illness.
4. The development of a means of summarising the range of risks appeared warranted which led to the adaptation of the Exceedence Probability method.
5. WAS disposal appears to be relatively effective in summer but less so in winter.
6. Summary statistics for particle abundance in sea water are informative by themselves.

**3.2.3. Secondary Summary Statistics**

Further summary statistics are found in Appendix 25 Summary Tables for Hydraulic Modelling Statistics. The tables have been derived from the MS Access table records of the number of particles, their mass, reduction and travel time in each scenario timestep record and summarise the following:

1. The timestep in each scenario when particles were first detected and its inverse statistic the % of timesteps where particles were detected.
2. The total number of timesteps out of 8516 for each scenario where particles have been detected;
3. The arithmetic average for all timesteps of the median travel times estimated at each timestep (in effect the typical travel time of the particles seen);
4. The minimum of the median travel times estimated at each timestep (in effect the shortest time in which pathogens can move from the outfalls to the beaches);
5. The average dilution experienced by pathogen particles detected at a bathing zone;
6. The average inactivation (as a decimal reduction) experienced by pathogen particles detected at a bathing zone;
7. The average overall reduction experienced by pathogen particles detected at a bathing zone.
8. The average of the total particle mass seen at each timestep (a relative estimate of the total number of pathogens reaching each bathing area over the 3 month model period).
Explanations of what each table shows are provided in the Appendix. When interpreting this data it is essential to recognise that particles disappear from the model either when:

- They are transported out of the coastal zone;
- Their mass is reduced below 1 unit by the solar inactivation;
- They are older than the maximum residence time allowed by the program.

This leads to some statistics that seem intuitively odd at first. For example the travel times under high inactivation are only of a few hours discussed above. Overall this data showed the same pattern as with Appendix 24 Summary of Hydraulic Modeling Outputs.

### 3.3. Assessment of Risk at Exposure Points under different Exposure Scenarios

The rest of this Chapter is designed to:

1. Further illustrate waste stream dilution and inactivation in the coastal zone (Section 3.4)
2. Quantify and report Infection and Illness Risk Probability (Section 3.6), in particular:
   a. Summaries of selected estimated risks in table form (Section 3.6.1);
   b. Risk estimates associated with ‘Baseline’ barrier performance not accounting for Hazardous Events involving on-shore transport of less diluted waste (Section 3.6.5);
   c. The situation of summertime risks under bright sunlight when there is a relatively high level of solar inactivation (Section 3.6.6);
   d. The effect of light reduction indicated by low light or conservative scenarios (Section 3.6.7);
   e. Winter related differences when coastal currents and tides and the exposed population could be expected to be different to those in summer (Section 3.6.7);
   f. The sensitivity of risk estimates to the input assumptions (Section 3.6.14) (includes surfers, more realistic radiation levels and cloudy days, higher than experimentally measured levels of pathogens, 2007 v. 2030 flows, disease levels in the community).

When this assessment was drafted some additional scenarios remained to be modelled. Nevertheless as can be seen in Appendix 24 Summary of Hydraulic Modeling Outputs and Appendix 25 Summary Tables for Hydraulic Modelling Statistics the data shows broad patterns which were evident in the initial set and used to draw critical conclusions (e.g. the main risk was from the 3 main pathogens for all inactivation conditions, 2007 v. 2030 were very similar), effluent was more important than WAS). This allow early conclusions to be developed and prioritising of Scenarios for modelling. One result was that modelling of WAS fate and transport was given priority over secondary effluent as the WAS disposal is more unusual and is of higher concern because of attributes being less well documented.

In the main text below illustrative risk exceedence probability statistics and plots and discussion of the findings are presented. A fuller set of risk estimates and plots can be found in:

1. Appendix 26 Exceedence Probabilities (summary statistics for exceedence probability modelling)
2. Appendix 27 Microbial Abundance and Risk Exceedence Plots I Nominal Dilution (plots of Baseline risks);
3. Appendix 28a Microbial Abundance and Risk Exceedence Plots II Secondary Effluent Summer (Baseline + Event Scenario) and Appendix 28b Microbial Abundance and Risk Exceedence Plots II Secondary Effluent Winter (Baseline + Event Scenario) (plots of Risks from secondary treated effluent incorporating on-shore transport events);
4. **Appendix** 29a Microbial Abundance and Risk Exceedence Plots III WAS Summer (Baseline + Event Scenario) and **Appendix** 29b Microbial Abundance and Risk Exceedence Plots III WAS Winter(Baseline + Event Scenario) (plots of Risks from WAS incorporating on-shore transport events).

Explanations of plot labelling can also be found at the beginning of the Appendices e.g. **Appendix** 27 Microbial Abundance and Risk Exceedence Plots I Nominal Dilution and **Appendix** 28a Microbial Abundance and Risk Exceedence Plots II Secondary Effluent Summer (Baseline + Event Scenario)

### 3.4. Variation in Waste Stream Dilution and Inactivation

Insight into pathogen fate and transport was gained by examining plots of the individual timestep data hydraulic data.

#### 3.4.1. Exceedence Plots of Particle Reduction

Exceedence Plots proved to be an effective way of presenting hydraulic data series. The Exceedence plots are illustrated in Figure 3-2. These graphs show the reduction under conservative and high radiation conditions. Further examples are presented in **Appendix** 30 Reduction and Dilution Plots From Hydraulic Modelling.

They showed how at percentiles above the 99th the reduction achieved fell below $10^3$ at all sites under conservative conditions in summer. Put another way instead of Baseline reduction of $10^5$ estimated for effluent under typical good conditions based on Beachwater water quality data could be reduced to only $10^2$-$10^3$. This occurred irrespective of the degree of solar inactivation assumed.

The limited impact of solar radiation was probably due to it only being effective for a short period of 4 to 6 hours each day, and this window allow rapidly moving particles to reach the bathing zone before being inactivated providing transport occurred during low light periods. Comparison of the conservative and high radiation scenarios also illustrated how:

- Effective solar radiation is at reducing pathogen numbers when it is active;
- A possible difference between Merewether Beach and Bar Beach on one hand and Burwood Beach and Dudley Beach on the other during summer with the former two being more of a concern.

Another observation was that the travel times to Merewether Beach and Bar Beach appeared substantially shorter than to the other two beaches even though they were slightly more distant (**Appendix** 25 Summary Tables for Hydraulic Modelling Statistics Table 4).
Figure 3-2. Reduction Factors (Dilution + inactivation) estimated from the hydraulic modelling in the case of effluent.

Notes:
1. Summer conservative (no inactivation assumed), b. Summer maximum inactivation.

3.5. The Episodic Character of Contamination Events

Exposure point timesteps characterised by limited dilution/inactivation did not occur randomly but were clumped or episodic (Figure 3-3). This was expected from the previous modelling (Glamore et al., 2008) which showed periodically plumes of less diluted discharge water moving into the shoreline waters and along the coast (see illustrations in Section 1.5.9). This observation is important for the following reasons:

1. It suggests a concurrence of one or more predisposing factor e.g. wind which might be predictable and this knowledge could be used for management e.g. warnings in the same manner as with rainfall and stormwater contamination;
2. It points to these reduced dilution/inactivation periods being definable as discrete hazardous ‘events’ rather than ‘hazardous circumstances’;
3. A more sophisticated analysis of risk and health impact potential is possible.

During summer these events were much more evident with the treated effluent than the WAS. With the conservative scenarios total reduction seen in the data was generally in the range of $10^3$ to $10^4$. 
noting that this may have been in part a modelling artefact of dilutions $> ca 10^4$ not being detectable. Nevertheless many spikes were so striking that the term ‘episodic’ seems reasonable. Also they contrast with the median dilution estimated of $>10^6$ (Glamore et al., 2008).

When inactivation was taken into account the range of reduction estimates increased greatly but the general episodic pattern was still evident. This episodic character supports the adaptation of traditional hydrological event exceedence plotting approach as the latter were designed with analogous episodic (rainfall) events in mind (e.g. Pilgrim & Doran, 1997).

A truism to remember in comparing these plots is that the different graphical formats used (e.g. Figure 3-2 and Figure 3-3) illustrate different event features. The exceedence plots highlight extreme/low frequency impacts, while the latter plots illustrate more typical event scale. Each style has its role in communicating and making decision relating to acceptable or tolerable risk (see Hunter & Fewtrell, 2001 for a detailed discussion of this issue). Figure 3-3 shows elevated risk. But it also highlights that even under Hazardous Event conditions the coast aquatic environment off Burwood Beach still provides several orders of magnitude protection against pathogen exposure. Figure 3-2 on the other hand shows the need to address the issue of events via formal risk assessment.
b. Figure 3-3. Distribution of timesteps corresponding to limited reduction in pathogen numbers due to dilution and inactivation over the 3 month modelling scenario period.

Notes:
1. a. Summer conservative (no inactivation assumed), b. Summer maximum inactivation.

### 3.5.1. Travel Times

Travel times for observed events were often shorter than one day. The effect was most pronounced under the high radiation scenario as this tends to remove particles below the model’s simulation limits.

Where a high level of radiation driven inactivation was assumed only pathogens travelling for less than 1 day were likely to be seen in the bathing zones (Figure 3-4).
Figure 3-4. Indicative Travel Times to Bar Beach for Effluent During Summer

Notes:
1. a. Summer conservative (no inactivation assumed), b. Summer maximum inactivation.
2. CountP4+ is the count of particles per timestep and indicates their relative numbers (4 particles is equivalent to ca a particle concentration in the discharge of 10^4)

3.5.2. Diurnal Timing of Events

Temporal distribution was not discernible in the pattern of dilution. Enhanced protection was associated with the time of day in summer when solar inactivation was high, however, this effect was slower and less pronounced in winter (Figure 3-5) when global solar exposure is only ca a third of mid summer and the UV component is even more reduced. The effect appears was more constrained still for the 15 MJ.m^-2 scenarios.

Overall the modelling indicated that solar radiation provides marked additional reduction of risk from late morning to later afternoon and confirmed its potential as a potent disinfection process for microorganisms persisting in the coastal zone for several days. However, its benefits are clearly likely to be constrained by the rapid travel time of constrained plumes, by the large number of people swimming in the morning, and by moderating factors such as poor water transmissivity and clouds. So it would seem unwise to rely on it to greatly reduce the risk posed by highly constrained and concentrated plumes when these move rapidly to one or other beach.
Figure 3-5. Variation in Particle Reduction (dilution + inactivation) for three contrasting irradiation scenarios

Notes:
1. Summer conservative (no inactivation assumed), b. Summer maximum inactivation. c. Winter maximum inactivation.
2. DF4+ = dilution factors for timesteps where at least 4 particles were found.
3. Reduction4+ = overall reduction factors for timesteps where at least 4 particles were found

3.5.3. Influence of winds and currents

Recognition of the episodic character of these hazardous water quality events associated with destratification + on-shore transport, and the potential risk to bathers implied, led to agreement that these events needed to be further characterised with a view to possibly improving their management.

To do this we first replotted the summer Conservative timeseries particle count data (Figure 3-6). This indicated the following:

1. The events were clearly ‘episodic’ by any commonsense meaning of the term;
2. Duration was typically less than one day though some longer events could occur;
3. There were also many gaps when modelling indicated that water would be of higher quality;
4. Events at Merewether and Bar were largely concurrent but they did not necessarily coincide with their counterparts at Dudley and Burwood.

In discussions WRL confirmed the likelihood that wind had a role but the effect was likely to be delayed, seasonal and be moderated by ocean currents as well. Accordingly a separate subproject was initiated to assess the correlation between event intensity and wind speed/run and direction (Pells et al., 2009). The aim of the project was at this stage to determine conclusively whether elevated contamination could be strongly linked to these factors to an extent that would warrant further work in support of event management e.g. early warning and monitoring.

Extracts from this work are presented in Appendix 31 Extracts for WRL Modelling of High Risk Periods. This study indicated that strong sustained on-shore winds are driving contamination but that the precise direction impacted on the degree of impact, this varied between the beaches and the contamination pattern may also vary with seasonal factors (compare wind directions in summer and winter corresponding to high event likelihood).
Figure 3-6. Timeseries Plots Showing (Microbial) Particle Intrusion into Surfing Areas
3.5.4. Event Frequency and the Problem of Acceptable Bathing Hazardous Event Risk

During the project planning and implementation process a range of questions were raised by HWC about high but infrequent e.g. concern whether the risks indicated reflect reality, where benchmarks should be set more generally. The critical issue seemed to be how to reduce define Hazardous Event risks and determine whether and when such risks were acceptable.

This was a concern for us, as defining risk benchmarks was seen as underpinning the format and structure of this risk assessment and recommendations regarding future management. The difficulty was that these benchmarks have not been satisfactorily identified in the Guidelines. The problem is raised here because our examination of the event hydraulic data showed clearly that Hazardous Events characterised by low dilution occurred every few weeks or days but their short duration meant they often might not be detected by standard Beachwatch monitoring, or at best there would only be intermittent signals of their occurrence.

This limited development definitions of hazardous events occurring relatively frequently was striking when contrasted with common practice in the fields of hydrology and drinking water quality where events with recurrence intervals of 100 years or less are considered worthy of management and events recurring at intervals of <1 year are considered ‘certain’ the highest likelihood category (e.g. EnHealth Council, 2002; NH&MRC/NRMMC, 2008; NRMMC/EPHC, 2005; Pilgrim & Doran, 1997; Standards Australia/Standards New Zealand, 2004a). Awareness of this ‘Best Practice’ was one reason we felt the need to estimate infection exceedence probability out to the 0.0001 level.

The latter was seen as approximating the risk posed by pathogens to a single person over a lifetime of bathing or one person among 10,000 beach goers acquiring an illness on any given day (‘exceedence probability’ is the correct way to express such risks but the latter form is noted here as though strictly incorrect it is current easier to conceptualise). Further discussion relevant to this matter is found in Appendix 14 Operational Application of Exceedence Probability Analysis To Hazardous Event Characterization.

This uncertainty about what should constitute and acceptable level of risk from bathing may be indicated by the following. is indicated by the WHO recreation guidelines(World Health Organization, 2003 p.2) in the following extract in their Executive Summary:

“The primary aim of the Guidelines is the protection of public health. The purpose of the Guidelines is not to deter the use of recreational water environments but instead to ensure that they are operated as safely as possible in order that the largest possible population gets the maximum possible benefit. The adverse impacts of recreational use of coastal and freshwater environments upon the health of users must be weighed against the enormous benefits to health and well-being—rest, relaxation and exercise—associated with the use of these environments.”

Regretably WHO does not indicate where the balance between hazardous event risk and relaxation should lie or suggest clear approaches to resolving the matter concretely. Further excerpts outlining the WHO position on balancing risk and relaxation are found in Appendix 33 Selected Extracts from the WHO Recreation Guidelines (World Health Organization, 2003).
3.6. Infection and Illness Risks Associated With Hazardous Events and Circumstances

Modelled estimates of infection and illness risk and potential levels to which bathers might be exposed are presented in this section:

- As summary statistics which indicate that the 0.01 illness probability benchmark proposed by NSW health is exceeded:
  - Commonly for enterococci (the measure of total gastrointestinal illness);
  - Periodically for Adenovirus and Giardia;
  - Much more markedly for surfers than the general population;
- As Exceedence probability plots which indicate how frequently contamination events occur under different scenarios and the scale of the impacts as functions of their likelihood.

3.6.1. Summary Percentiles

Summary risk statistics are shown in Table 3-1, Table 3-2, Table 3-3 and Table 3-4. Risks are to increase/be higher for:

- Surfers (This is due to the assumption that they consume on average ca 7 times the seawater volume of a normal bather);
- Low light conditions (solar disinfection will be ineffective);
- Winter conditions compared to summer (WAS appears more likely to surface in winter compared to summer).

Risk was most pronounced for ‘enterococci’ in its role as a surrogate of all gastrointestinal pathogens. The rise in risk and its significance appears rapid which can be accounted for by:

- the steep slope of the empirically derived dose response algorithm (Kay et al., 2004; Kay et al., 1994) which is only sensitive to the higher exposure levels unlike more normal algorithms (exponential, Beta Poisson);
- the fact that this parameter aggregated risk for all pathogens;
- limitations in the pathogen assays.

3.6.2. Baseline

We estimated that if Baseline assumptions prevailed all the time it can be seen that risk from the water to which shoreline bathers would be exposed would be tolerable up to the 99.95th percentile for Exceedence Probability.

Campylobacter can be seen to pose the least risk. Cryptosporidium tends to pose less risk than Giardia.

3.6.3. Effluent Under Baseline+Event Conditions

For shoreline bathers, risk as exceedence probability was 0.01 ca 5 to 8% of the time for the 15 MJ.m⁻² summer scenario (2030). This is in line with the risk survey underpinning the Guidelines.

For surfers, risk as exceedence probability was 0.01 ca 10 to 20% of the time for the 15 MJ.m⁻² summer scenario (2030). The same level of risk appeared to exist from effluent during winter with the possible exception of Merewether Baths.
Risks at the different locations were very similar with Bar and Merewether being typically the same where they were directly comparable e.g. summer effluent Conservative and 3 MJ.m\(^{-2}\) Scenarios. Burwood, Dudley and Merewether were similarly comparable under the 15 MJ.m\(^{-2}\) Scenarios.

3.6.4. WAS under Baseline+Event Conditions

Risks are much more pronounced in winter than in summer and are not substantially greater generally than for secondary effluent.

Risks from WAS were well below those posed by effluent particularly in summer. They also tended to be lower in winter by a factor of \(ca\ 5\). This is in line with the different contribution to the loadings from effluent v. WAS.
<table>
<thead>
<tr>
<th>Simulation Type</th>
<th>Waste stream</th>
<th>Beach Population</th>
<th>Season</th>
<th>$S_{90}$</th>
<th>Pathogen</th>
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3.6.5. Plots of Baseline Risk to Normal Bathers from Secondary Effluent + WAS

These figures also indicate that when the pathogen numbers in the effluent and WAS are reduced to a normal degree they pose a tolerable risk (<1 %) to normal bathers irrespective of the variance in the levels of the pathogens at the discharge point.

This is consistent with the current normal (95th percentile) classification of these beaches as class A microbiologically.

![Graphs of Baseline Risk](image)

**Figure 3-7. Exceedence Probability Plots for Risk Assuming secondary effluent diluted by 10^5**
3.6.6. Plots of Baseline+ Event Summer Risks (Best Dominant Conditions for Shoreline Bathers)

Given median exposure conditions (Baseline + periodic on-shore transport events 15 MJ.m$^{-2}$ ) risk to normal shoreline bathers from effluent was comparable to or better than the 1% benchmark at the 95th percentile (Exceedence prob.= 0.05) (Figure 3-9). The risk from WAS was still lower (Figure 3-10).
WAS plots appeared to notable for on-shore transport and hence conditions of markedly elevated risk being marked but rare. The ‘step’ seen commonly in the plots reflect the bimodal nature of the Baseline + event PDFs discussed in Appendix 14 Operational Application of Exceedence Probability Analysis To Hazardous Event and the specific attributes of the two or more underlying distribution.
Figure 3-9. Exceedence Probability Plot for Pathogen Risk for Selected summer 15 MJ.m\(^2\) Scenarios and Shoreline Bathers
3.6.7. Winter Risk

The winter risk situation for effluent appears similar to that in summer (Figure 3-11) except that there appeared to be more on-shore transport of WAS and increased risk as a result (Figure 3-12). The step increase was very pronounced because of the higher level of contaminants in the discharge.
particularly in the case of *Giardia*. From the modelling the step change appears at an exceedence probability around 0.02 (98th percentile).

Figure 3-11. Exceedence Probability Plot for Pathogen Risk Assuming Baseline secondary effluent dilution by $10^5$, Winter Conditions and High Radiation
Figure 3-12. Exceedence Probability Plot for Pathogen Risk Assuming WAS Baseline dilution by $10^6$ for Winter Conditions and High Radiation driven inactivation
3.6.8. Risk from ‘All Pathogens’ (as Enterococci) v. Index Pathogens

Modelling was used to estimate the risk arising from all pathogens collectively based on enterococci levels and the risk from selected index pathogens. The highest risks were seen with the ‘all pathogens’ group for both effluent and WAS indicating that the enterococci dose response curve was conservative. The risk patterns estimated by normal QMRA using the two main index organisms (Giardia and Adenovirus) were generally in line with those seen identified using enterococci but were lower as would be expected for a subgroup of pathogens.

During the project planning phase it was mooted that risks from individual pathogens might use an illness probability of 0.001 as a benchmark. The exceedence probabilities where this occurred can be seen in:

1. Appendix 26 Exceedence Probabilities (summary statistics for exceedence probability modelling);
2. Appendix 27 Microbial Abundance and Risk Exceedence Plots I Nominal Dilution (plots of Baseline risks);
3. Appendix 28a Microbial Abundance and Risk Exceedence Plots II Secondary Effluent Summer (Baseline + Event Scenario) and Appendix 28b Microbial Abundance and Risk Exceedence Plots II Secondary Effluent Winter (Baseline + Event Scenario) (plots of Risks from secondary treated effluent incorporating on-shore transport events);
4. Appendix 29a Microbial Abundance and Risk Exceedence Plots III WAS Summer (Baseline + Event Scenario) and Appendix 29b Microbial Abundance and Risk Exceedence Plots III WAS Winter (Baseline + Event Scenario) (plots of Risks from WAS incorporating on-shore transport events).

During summer under sunlit conditions the illness risk to shoreline bathers from the Adenovirus and Giardia were always <0.001 per exposure at an 0.01 Exceedence Probability level (99th percentile) for effluent and WAS. Risks however showed exceedence probabilities markedly higher than 0.01 with surfers, under conservative inactivation assumptions and/or with WAS in winter. This was comparable to the risk pattern seen with enterococci.

3.6.9. 2007 v. 2030

The central driver for this risk assessment was the proposal to expand the STP. Comparison of 2007 and 2030 scenario statistics indicated the increase in risk from both effluent and WAS would be approximately proportional to the increase in flows (ca 20% and 40%) or 0.1 and 0.15 log10 units respectively. This is so small compared to other uncertainties and variance associated with the risk estimates that it is arguably a secondary consideration in for plant redesign decision making. This lack of impact on risk can be seen in WAS simulations in Table 3-4.

3.6.10. Seasonality

Per person risks can be seen to be higher in winter probably because of:

1. Lower disinfection;
2. A greater tendency for WAS to surface.

This increase would be mitigated on a population basis by fewer bathers using the waters during these months.
3.6.11. Between Beach Differences

The beaches were similar in risk magnitude. A possible slight tendency for Merewether Baths and Bar Beach to be worse that Dudley or Burwood Beach though this could be a reflection of the assumptions used (e.g. environmental conditions for 3 months assumed to be general).

3.6.12. Source Dominance Effluent or WAS?

As a general rule for equivalent Scenarios pathogens numbers from the effluent were higher that those from the WAS and as a result dominated risk for any give probability. The main exception was for WAS during winter when Giardia dominated (Table 3-4).

The two streams were not combined in the risk estimations because no further useful additional information would be obtained thereby. The dominance of one source or another meant that at worst if the two loads were identical the additional risk from the subdominant source would increase the total estimate by at most 0.3 log₁₀ units. In practice the loads were not coincidental and the risk estimates varied over many logs. So the effect of adding the risks would be at best trivial and at worst given the impression that total risk is known with such precision that a change of 0.1 to 0.3 log₁₀ units is critical for decision making when there are many other uncertainties which are more likely to affect risk estimates.

Partly because of such uncertainties we have compared risk estimates against the benchmark 0.01 gastrointestinal illness probability in bands of ca 0.3 log₁₀ units as can be seen in Table 3 2 and Table 3 4 and estimated risk at intervals of approximately this magnitude.

3.6.13. Effect of Solar Radiation

When solar radiation is assumed to have no effect, risks are elevated somewhat compared to the illuminated conditions though not enormously so. This is most likely because of:

- The short travel times associated with many events;
- That the assessed risks covers both day and night hours.

This fact needs to be borne in mind when interpreting the risks estimates for conservative scenarios. However they cannot be completely discounted because:

- The scenarios shown here are for best case inactivation rates;
- People often swim at dawn or in the morning before solar radiation is likely to have had an impact;
- Overcast, autumn and winter days are less protected by Solar radiation.

To gain further insight the 15 MJ.m⁻² scenarios were modelled. As expected they were intermediate between the high light intensity and the conservative scenarios, generally close to midway.

It is suggested that the 15 MJ.m⁻² scenario risks are the first choice criteria for reporting and decision making. However, bearing in mind the range of known uncertainties, the conservative scenarios risk estimates should also be considered in a ‘weight of evidence’ rather than ‘compliance’ style use of the risk estimates.
Figure 3-13. Exceedence Probability Plot for Pathogen Risk Assuming Baseline secondary effluent dilution by $10^5$ but With no Solar Radiation Driven Inactivation

The above plots provide an overall risk picture. However risks may be elevated beyond these for a range of reasons. This section considers using sensitivity analysis the extent to which risk may be higher or lower depending on the assumptions in the modelling.

In addition to the plots shown here see also Appendix 32 Sensitivity.

3.6.14.1. Surfers

A question of concern to the CRG was whether surfers had a higher level of risk compared to other beach users. Reportedly surfers are relatively more active during winter than the average bathers. As a result elevated risk from winter levels of pathogens needed to be assessed.

The estimated risks seemed to support these concerns in that:
1. The risk to surfers was higher generally by virtue of their high assumed intake;
2. The risks estimated as a result more often exceeded the 1% benchmark;
3. Use of the beaches in winter would expose surfers to WAS pathogens to a much greater extent than was the case with the normal population in summer;
4. Above the 99th percentile in winter the illness risks from individual pathogens exceeded the 1% benchmark.
Figure 3-14. Exceedence Probability Plot for Pathogen Risk Assuming Baseline secondary effluent dilution by $10^5$, Summer Conditions and High Solar Inactivation

Figure 3-15. Exceedence Probability Plot for Pathogen Risk Assuming Baseline secondary effluent dilution by $10^5$, Winter Conditions and High Solar Inactivation
Figure 3-16. Exceedence Probability Plot for Pathogen Risk Assuming Baseline WAS dilution by $10^6$, Winter Conditions and High Solar Inactivation
3.6.14.2. Underestimation of Adenovirus Level

Adenoviruses were detected in comparable numbers to those reported in the literature for cultural assays but well below the numbers reported from PCR based assays (He & Jiang, 2005) by a factor of ca 100. So the simulations were rerun assuming initial levels in effluent 100 X those detected by cultural methods.

The result was that Adenovirus illness risk under event conditions exceeded 10% at the 0.01 probability level. This is notable as it suggests one way to account for the high overall ‘enterococci’ risk estimates under event conditions along with Norovirus (see Appendix 10 Seasonality, Outbreaks and Hazardous Events).

![Figure 3-17. Exceedence Probability Plot for Adenovirus Risk Assuming Baseline secondary effluent dilution by 10^5 + Event Risk Indicated by Scenario](image)

**Notes:**
1. a. Reference conditions, b. Assumption that true Adenovirus level is 100 X higher than estimated by cultural methods
3.6.14.3. Underestimation of Giardia

Recovery of spikes of Giardia and Cryptosporidium were generally < than the recommended 10% suggesting the assay was not optimised.

To test if poor recovery could influence risk estimates we reran the summer secondary effluent scenario assuming Giardia numbers were in fact 10 X greater. This showed that at illness risk increased to a level of concern such that the 1% benchmark was exceeded at the 99.9\textsuperscript{th} percentile. The 0.1% infection probability was exceeded at the 98\textsuperscript{th} percentile.

Set against this was uncertainty about the viability of the Giardia cysts as the adjusted counts were total counts rather than confirmed or viable counts.

![Giardia Exceedence Probability Plot](image)

**Figure 3-18. Exceedence Probability Plot for Giardia Risk Assuming Baseline secondary effluent dilution by 10\textsuperscript{5} but assay efficiency gave only 10% of the true number**

3.6.15. Seasonality and Outbreaks

3.6.15.1. Rate of disease in the Community

An issue raised by NSW health was whether the levels of gastrointestinal disease in the community might at times be higher and lead to elevated risk as a result beyond those estimated. To estimate the possible impact we examined the rates of gastrointestinal illness in the NSW community (Communicable Diseases Branch, 2008) for 2007. Based on the latter reference it appears that monthly disease rate was no more than 3 X higher than that in September to November for the following major types of illness/infectious agent which could be acquired from diluted effluent:

1. Cryptosporidiosis;
2. Giardiasis;
3. Institutional gastroenteritis;
4. Salmonella infection;
5. Hepatitis A

This is a relatively small change compared with other discussed above such as for Adenovirus and Giardia. And Cryptosporidium appeared to be a lesser concern.
A limitation of this study was that gastrointestinal disease prevalence can be seasonal (e.g. Institution Gastroenteritis Table 3 Communicable Diseases Branch, 2008) and might lead to peaks associated with outbreaks (e.g. Lee et al., 2001).

The problem of seasonality and outbreaks appears at present too complex and intractable to be addressed here in a useful fashion. The issue is reviewed and discussed in Appendix 10 Seasonality, Outbreaks and Hazardous Events and Section 2.6.5. Special sensitivity testing could be done for agreed scenarios but:

1. Such estimates would necessarily be very speculative at present;
2. They would not change the beach water quality ratings;
3. From the sensitivity assessments above we already know that a relatively small level increase of 1-2 orders of magnitude in a dominant pathogen will likely making infection very likely.
4. The literature provides possible values but it does not indicate which are appropriate. From the data collected by HWC, Cryptosporidium and Giardia here were comparable to the numbers seen in polluted rivers under storm conditions in Australia (Roser & Ashbolt, 2007), southern highland WWTPs (SCA internal report prepared by UNSW), in sewage in the Netherlands (Medema & Schijven, 2001) and Canada(Payment et al., 2001). But these levels were 1 to 2 log\(_{10}\) units below those reported by others in the US (Madore et al., 1987) under normal circumstances, and less still than in an outbreak in a small communities (Lee et al., 2001). Overall these data suggested the level in effluent might be one to two logs higher in the waste from a large city like Newcastle so it is not clear that it is applicable.

As a result, particularly bearing in mind the last point, it is suggested in the absence of additional information, the sensitivity analyses carried out above for Adenovirus and Giardia provide an indication of outbreak impacts on swimming risk.

3.6.15.2. Assumed Baseline

As previously discussed the Baseline beach water quality was consistent with reductions of ca \(10^5\) (treated effluent) and \(10^6\) (WAS). The question arises as to whether reductions particularly for WAS might be less than this. To assess this we estimated the beach water quality in terms of enterococci if the reduction in WAS pathogens was only \(10^5\). If this were the case then much higher levels of enterococci would have been detected than were observed in the monitoring program. If this higher level of contamination occurred then the median enterococci levels seen would have been ca 10 per 100 mL rather than the actual dry weather median of < 1 per 100 mL.

A point to note related to this is that the modelled Baseline risk estimated using enterococci reduction assumptions is very sensitive to the assumed Baseline reduction value (e.g. \(10^5\) in the case of treated effluent), while the event based levels and risks are more clearly grounded.
Figure 3-19. Exceedence Probability Plot for ‘Enterococci’ Risk Assuming Baseline WAS dilution was only $10^5$
4. Key Findings

1. The results of the risk assessment (pathogen numbers, illness risk) are specific to the input assumptions used to construct the exposure pathway based scenarios. It is emphasized that the risks are modeled ones developed to support decision making by project stakeholders. They should not be confused with actual risks of disease calculated via epidemiological studies.

2. The study found that under typical conditions local beaches have very good quality consistent with Beachwatch results. However, the study also found that under infrequent hazardous conditions the discharge of effluent and, to a lesser extent WAS, does impact on the beaches and has the potential to present an elevated health risk to local beach users during these periods.

3. Under Baseline conditions the infection and illness risk probability, modelled for all pathogens (enterococci) and individual pathogens (Adenovirus, Giardia in particular) at the 95th percentile (Exceedence probability 0.05), were generally < 0.01 (i.e. 1%) for summer shoreline bathers. This was consistent with Beachwatch enterococci monitoring which indicated the beaches achieved Category A water quality status (infection prob. < 1%) based on the 95th percentile of the enterococci measurements taken during dry weather.

4. The study also provided detail of the risks associated with ‘Hazardous Events’ when there was sporadic, short lived on shore transport of outfall discharge material. This was found to be closely associated with a combination of destratification of the water column, on-shore currents, and strong or extended duration on-shore winds.

5. Under Hazardous Event conditions:
   a. The risk estimates were highly dependent on the risk exposure scenario assumptions in particular season, discharge stream, solar inactivation and bather populations.
   b. For shoreline bathers the elevated gastrointestinal illness risk (probability >0.01) was assessed to occur at an exceedence probability of 0.05 to 0.08 on sunny days.
   c. Under a worst case scenario (no sunlight) for surfers ingesting 200 mL of seawater per occasion the elevated gastrointestinal illness risk (probability >0.01) was assessed to occur at an exceedence probability of 0.2 to 0.5.

6. Surfers appear to be the higher risk group as:
   a. It was assumed they undertook more vigorous and extended bathing and would consume 7X the seawater of normal bathers per occasion;
   b. Their use of the ocean in winter and early morning could lead to higher risk due to different contaminant transport patterns and solar inactivation being less effective.

7. Solar inactivation can be an effective moderator of risk during daylight hours. However, it was less effective than might be expected because of short travel times (often less than 1 day) combined with solar inactivation occurring mainly around midday.

8. Events of poorer quality water were episodic and could occur at any time of the day.

9. For the most part the study found that the effluent discharge presented a larger health risk than WAS.

10. Microcosm experiments found that following dispersion WAS and effluent showed similar inactivation rates which were also comparable to rates reported in the literature.

11. The study found that assessing the health risk based on predicted enterococcus levels did not underestimate the potential health risk estimated for individual viral, bacterial or protozoan pathogens in the effluent and WAS discharges from the outfall. On this basis enterococcus appears to be a satisfactory indicator for Burwood effluent and WAS discharges under normal circumstances. A possible exception would be if there was a large outbreak of disease in the community. Of the index pathogens assessed Adenovirus and Giardia were the most significant.
12. The different beaches and swimming locations did not appear to differ greatly from one another in the levels of pathogens seen during a given season.
13. The impact of the Plant upgrade and larger discharge volumes anticipated over the time seems very small compared to the other sources of risk variability and uncertainty.
5. Uncertainties

5.1. Model output prediction

The illness risk estimates presented here are based on QMRA theory and explore risk options. They are designed to understand reality and estimate the level of risk based on the use of reasonable assumptions. But they are still model outputs. One concern raised was whether a relationship between enterococci numbers and gastrointestinal illness developed in England and Wales for Newcastle Coastal water. For example, it is unclear what the source of the enterococci pathogens detected in the epidemiology survey was and how they compare with those discharged by BBWWTP. The assumed enterococci dose response curve given the concern is human sources is reasonable but it is still unproven in the Australian context.

5.2. Reliability of the Surfer sea water consumption (200mL per exposure)

This figure was based on qualitative assessment. However, it is comparable to the normal maximum for chemical contaminants identified in the guidelines so it seems credible as a worst case assumption.

5.3. Consumption of seawater in a single timestep

In practice seawater consumption might occur in more than one timestep. The impact of this could be simulated with further work but the in the authors’ opinion would not change the overall risk picture markedly.

5.4. Clustering of hazardous (timestep) periods

Particle rich timesteps were clearly clustered in the hydraulic model outputs. This would not effect the infection risk probability if it is assumed that the majority of accidental seawater ingestion occurred within a 15 minute period e.g. as a result of a sudden large volume consumption after being rolled by a breaker and being ‘winded’.

To resolve whether this makes a maked difference this further risk assessment might be undertaken after the following actions provided more data:
1. A survey of bather swallowing of seawater;
2. Comparison of risk if it is assumed that consumption occurred in 2 or more sub-events.
   (This would need to take into account the fact that water contamination appears to occur in extended periods which are close in time).

5.5. Assumed Level of Baseline Protection

The $10^5$ and $10^6$ assumptions for effluent and WAS were estimated from the long term enterococci measurements. Modelling might be used to assess how realistic these are. Better data on this might be obtained by surveying a relatively abundant and conservative contaminant component found in effluent or WAS such as faecal sterols (Leeming et al., 1998).

5.6. Number of Monte Carlo Iterations

The number of Monte Carlo iterations per model run was set at 10,000 because of the available time and the number of simulations needing to be done. It may be that these may have led to some
underestimation of the extreme exceedence probability risks. This could be addressed by redoing
selected scenarios of high concern with a larger number of iterations (n=100,000).

5.7. Data Set Numbers and Quality

A concern noted by the assessment reviewer was the size of the data sets used to develop the PDFs.
We strongly concur that the larger the sample size the better. The matter was raised as an issue in
discussions NSW Health and DECC. The final number chosen of 12 samples + 6 controls at each
point in the end was a value judgement of what constituted in our experience sufficient data to
estimate ‘typical’ pathogen levels over the 3 month survey period agreed to and what was in
practice feasible.

Our decision was not based solely on statistics but rather on a number of considerations. Further we
did not see it as necessarily a final data set an initial collection designed to concurrently collect a
reasonable quantity of pathogen data and test the capacity of local analytical resources to undertake
the task within the short assessment timeframe proposed.

The question of how many samples is a vexed one in the field of environmental, especially
pathogen microbiology, because the variance is large. The Guidelines in line with WHO
assessments ideally recommend 20 measurements of indicators in bathing water over a season
ideally and collection of measurements over 5 years at each sampling point. This level of
monitoring has been applied in NSW to the Beachwatch program and has led to the accumulation of
large indicator data sets. The need in part arises from the goal of being able to estimate the 95th
percentile reliably and the recognition of the need to detect if possible high risk periods and include
their occurrence in the water quality data sets.

The application of such a principle to pathogens though is more difficult. The reasons are inter-
related and as follows.

There are as yet no guidelines, or as far as we know a consensus, on the number of pathogen
samples should be collected from sewage for QMRA of bathing waters. The Guidelines were
developed with indicators instead of pathogens in mind, with exposure points rather than ‘source
waters’ in mind, and the recommended timeframe for collection of five years was judged too long.

Statistical theory was seen of limited use in identifying how many measurements would be required
for decision making regarding the STP upgrade, other than to say that more is better. But this
principle does not provide a precise answer. The current climate change debate illustrates the
problem. The principles of global warming and its effects have been known for 100 years but there
are still continued calls for ‘more data’ to ‘satisfactorily understand’ climate change. One problem
is that the more data that is collected the more uncertainties and new questions tend to appear and a
circular logic develops which if applied in a pure form would prevent any decision being made.

In reality a 100 sample target is itself a small set and is in fact itself ad hoc. Haas has discussed this
problem and noted that detecting the high pathogen level events probably needs more samples still
because of their relative infrequency, and indeed unknown frequency, of ‘hazardous events’ (Haas,
1997).

Then there are resource constraints. Part of the problem is the perceived cost of pathogen
monitoring. At ca A$500 per assay on top of collection costs managers strongly question the
cost/benefit of pathogen assays and open ended statistical discussions are generally insufficient.
Because of this the logistic and human resources actually available to do the assays in the past those
currently available tend to be limited and at times of uncertain value. The result is that even where financial resources are available for additional testing, as in a crisis, testing cannot be easily expanded in any case. Put another way our survey needed to account for laboratory resources available to do the work with at least their normal proficiency. As a result our decision on how many samples to collect was based as much on management and ‘best practice’ considerations as statistics and guidelines.

The scientific literature by default arguable might have provided a benchmark. As part of a separate study we have been examining a range of sewage and sludge surveys and these informed our sampling in a general manner. Illustrative examples are as follows:

1. Ottoson and colleagues assayed Cryptosporidium and Giardia, Norovirus and enterovirus in 2 to 23 samples per waste stream (Ottoson et al., 2006).
2. Medema and Schijven in their survey of STPs analysed 31 samples per wastestream type for Cryptosporidium and Giardia (Medema & Schijven, 2001) but these appeared to be across 5 STPs suggesting 6 samples per sampling point.
3. Bosch et al. analysed a total of 15 rotavirus and enterovirus sampled from different parts of two STPs (Bosch et al., 1988a).
4. Madore only analysed 4 sewage and 11 sludge samples from different points (Madore et al., 1987).
5. A Campylobacter survey in the Netherlands analysed 16 to 30 single samples in and out of 3 STPs (Koenraad et al., 1994). Notably the variance in their numbers was high as with ours study.
6. A Dutch survey (Lodder & de Roda Husman, 2005) sampled STP viruses on only 5 occasions over a single season. Repeat analyses were undertaken but it was unclear how many replicates were analysed. Assuming 3 per occasion this would amount to 15 samples per sampling point.
7. No numbers were identified by Lee et al. but they appear to be small (Lee et al., 2001)
8. Petrinca et al. analysed 96 sewage samples for a range of viruses but this amounted to only 4 samples per point per STP (Petrinca et al., 2009).
9. Another study analysed a total of 29 sewage samples for rotavirus (Mehnert & Stewien, 1993).
10. Pusch et al. conducted a larger study of some 51 virus samples immediately downstream of Leipzig STP (Pusch et al., 2005).
11. The most extensive study was 80 samples (Payment et al., 2001) for protozoa and viruses but the data was somewhat problematic with unaccountable consistency (same level 8 days in a row!) in the numbers detected beyond anything but no information on sample reproducibility or much other critical QC information.

What this indicated was six-fold:
1. There is remarkably limited data on pathogen levels in sewage and this is even more limited when the range of different treatment processes is considered.
2. None of the studies we identified approached the 20 samples per year over 5 year WHO model.
3. Survey philosophy ranges from studying a single sample type in much detail to a broad reconnaissance survey.
4. Many surveys concurrently collected many more indicator measurements suggesting they too were resource constrained and hence the sample sizes were like ours a pragmatic compromise.
5. Few surveys collected a full range of pathogens from viruses, bacteria and protozoa.
6. Overall our survey scale of 54 samples was in fact quite comparable to this ‘best practice’.
Counting the diurnal samples 30 enterococci measurements were made of the latter’s abundance. In the case of the pathogens we not only collected 12 samples per sampling point but also another 6 control samples. The latter is noteworthy as few of the studies above noted the results of any control samples and the use of statistics was limited.

What level of variance is needed for QMRA is not even clear. For example Gale assessing sludge used simple point estimates direct from the literature to understand system risks to satisfactory degree (Gale, 2005a).

5.8. *Campylobacter* Levels and Sensitivity Testing

We were concerned that *Campylobacter* levels might be underestimated. However it is unclear at this stage what would be an appropriate sensitivity factor to employ given the uncertainties associated with its levels in sewage (Section 2.3.1.5).

5.9. Water Transmissivity

The inactivation and transmissivity study (Appendix 16 Inactivation Studies Covering Microcosms and Water Transmissivity Experiments) showed that water might often be much less transmissive than was modelled. For this reason we recommended considering risks estimated for both the 15 MJ.m$^{-2}$ and the Conservative scenarios. Because the survey was only of limited duration (one day) it cannot be considered to fully represent water transmissivity overall.

A more sophisticated model might take account of decreased water transmissivity associated with limited effluent and WAS dilution. However this would need to consider the effect on the full water column and would be unlikely to yield additional good data.

5.10. Increase in Illness Probability due to Seasonal and Outbreak Related Pathogen Peaks

Further QMRA could conceptually have been undertaken to detail the impact of seasonal and outbreak related pathogen peaks. Because seasonality and outbreak impacts are both largely unquantified, or where detected, they have a very high effect, a minimum sensitivity factor which should be considered is 1 or 2 orders of magnitude as already been done for *Giardia* and Adenovirus above (Section 3.6.14). However, it is not clear what inputs assumptions should be used and it did not appear that such an exercise would generate anything useful beyond what can be deduced from first principles.

This was because Conservative model sensitivity assumptions were seen as likely generating predicted illness risks approaching unity. The likelihood of this was evident from first principles and from the following simple modifications to the Scenario simulations already undertaken. Firstly there is the outcome of assuming that the risk of total gastrointestinal illness increased by 10 i.e. one order of magnitude. This increase would be equivalent to assuming enterococci numbers at the beach were increased by a factor of 10 or the Baseline barrier effect of the coastal waters was decreased by a factor of 10. Increasing enterococci numbers by a factor of 10 leads to the 95th percentile for enterococci being increased from its present values of ca 15 – 40 .100mL$^{-1}$ to 150 - 400 .100mL$^{-1}$ with the result that depending on the preferred way of calculating the 95th percentile the beaches would be considered water quality category B or C for normal bathers and category C to D for surfers.
Similarly if we assume a reduced Baseline barrier effect equivalent to 1 order of magnitude the effect together with the increased surfer consumption on exceedence probability plots is shown in Figure 5-1. It can be seen that risk of illness/infection exceeds 0.01 at the 50th percentile for both effluent stream and the WAS, and particularly for the WAS. The effect of such sensitivity analysis can also be seen by considering how the enterococci risk curves in Figure 3-14, Figure 3-15 and Figure 3-16 would be affected by an increased risk of *ca* 1 order of magnitude.

Conservatively such sensitivity analysis would likely lead to one of two conclusions:
1. Any substantial outbreak or marked seasonable peak could make bathing water at the Newcastle Beaches exceed the 0.01 probability benchmark during the time of the peak and should be seen as hazardous. In this case NSW Health and Beachwatch need to consider how they are to scope and monitor such events in the community as well as manage beach access;
2. Sensitivity analysis is at this stage too speculative for reasons discussed in Appendix 10 Seasonality, Outbreaks and Hazardous Events.

In regard to the latter point the following is noted on page 27 of the Enhealth Guidelines (EnHealth Council, 2002):

> “Risk assessment is inappropriate when it is a ritual rather than a meaningful process and should not be undertaken when:
> • there is no data or an insufficient amount of data;
> • there is an inability to take action or it is too late to take action;
> • there are insufficient resources; and
> • it is politically or socially unacceptable.”

We would contend that beyond the simple analysis we have done, risk assessment of seasonality and outbreaks is currently in the first category and in our opinion the second conclusion is preferred.
Figure 5-1. Baseline Risks assuming enterococci numbers are 10X higher than is indicated.

Notes:
1. 1-percentile = exceedence probability.
2. For further explanation of labelling see Appendix 27 Microbial Abundance and Risk Exceedence Plots I Nominal Dilution (i.e. Baseline) and Appendix 14 Operational Application of Exceedence Probability Analysis To Hazardous Event Characterization.

5.11. Accommodating uncertainty

Uncertainty assessment, reality checks and related work were integrated into the general study design in a number of ways:
1. Communication and reporting has been designed to highlight that risk cannot be reduced to a single value but:
   a. is scenario specific;
   b. risk probabilities themselves for a continuum;
2. Multiple scenarios have been simulated so that decision making can be based on a weight of evidence;
3. Data from each model run has been compiled in database table format before being used in modelling to allow efficient checks for data integrity.
4. The assessment of multiple scenarios covers:
   a. Between population variation in risk;
   b. Between beach variation in risk;
   c. Between pathogen variation in risk;
   d. Between waste-stream type;
   e. Risk associated with different reduction/inactivation/dilution scenarios;
   f. Effects of seasons of risks;
   g. Effects of increase loading/different year on risk;
   h. Comparison of primary risk estimates with those when credible differences are possible (sensitivity testing);
5. Selection of the Baseline scenario was based on assessment of the consistency of observed water quality with that which would be expected;
6. Literature has been used to assess and identify possible underestimates of pathogen levels and scenarios where sensitivity testing should be undertaken;
7. The project plan was reviewed by NSW Health and DECC and the output results will be as well;
8. The hydraulic models themselves are probabilistic and:
   a. they generate aggregated statistics reflecting between timestep variation;
   b. they generate statistics of contaminant particle behaviour in the coastal waters relating to time of travel to the beach, degree of inactivation, and clustering of
contamination consistent with plumes of material periodically reaching the swimming zones;

9. The risk assessment overall is based on the principle of knowing (defining quantitatively) the system as well as possible;

10. The oceanographic conditions were compared with longer term data sets to assess whether they are representative;

11. Experimental checks have been used to assess
   a. Typical inactivation rates in the waste streams;
   b. Compare WAS with effluent and determine;
   c. Determine whether inactivation of different model microorganisms is consistent with literature values and one another;
   d. How effective solar radiation is likely to be in the ocean based on measurement of the transparency of water samples.
6. Conclusions

6.1. Primary Conclusions

The following are the central conclusions of this assessment.

1. Episodic periods of increased risk (i.e. >1% gastrointestinal illness probability) do appear to occur.
2. Surfers were assessed to be a population at higher risk because of their more vigorous and prolonged exposure to seawater and hence likely higher intakes.
3. Water quality degradation appears to be due to both the treated effluent and to a lesser extent WAS discharges. The higher pathogen numbers in WAS are off set by its smaller volume.
4. Following dilution solar radiation can effectively inactivate microorganisms from WAS and effluent to closely comparable degrees at rates comparable to those reported in the literature.
5. However solar radiation’s protective effect is constrained by short travel times (often less than 24 h) of some plumes during onshore transport events, and during low light periods e.g. early morning.
6. Compared to other sources of risk variance and uncertainty the impact of the proposed Plant upgrade appears small if not trivial.
7. The impact of disease outbreaks in the community and the seasonality of pathogen loads remain for the moment unresolved.
8. The results of the risk assessment (pathogen numbers, illness risk) are specific to the input assumptions used to construct the exposure pathway based scenarios. It is emphasized that the risks are modeled ones developed to support decision making by project stakeholders. They should not be confused with actual risks of disease calculated via epidemiological studies.

Follow-up work has not been fully scoped though a preliminary risk provided in Appendix 35 Follow-up Work.

6.2. Special Concerns

In regard to the following issues of concern we concluded:

1. Do the solids loading in the WAS pose a problem through shielding pathogens within their matrix? Do viral pathogens present an elevated health risk to beach users? Are enterococci a satisfactory indicator of risk and viral pathogens?
   a. Once the WAS has been diluted by a factor $>10^3$ the microorganisms contained therein are as susceptible to inactivation via solar radiation as those in the effluent, at rates in line with those reported in the literature;
   b. The enterococci ‘surrogate’ dose response curve appears to provide a conservative estimate of total gastrointestinal illness compared to illness risks estimated for individual pathogens, even allowing for pathogen assay limits. This included the index virus chosen Adenovirus.
2. Did the WAS pose a risk higher than that arising from the effluent?
   a. The WAS appears to pose in general a lower risk than the effluent especially during summer.
3. How does the current work compare with outputs from the Beachwatch monitoring program?
   a. Beachwatch addresses the primary question of overall water quality for shoreline bathers.
b. Beachwatch data is able to detect the occurrence of Hazardous Events via elevated indicator levels, notably stormwater inflows, but not characterize them in full detail.

c. Beachwatch indicator monitoring data can to detect other events by inference based on high levels of enterococci in absence of heavy rainfall. However such monitoring data alone is insufficient per se to distinguish the individual sources e.g. on-shore transport of outfall discharges, bather shedding and sediment suspension.

d. The current study complements Beachwatch monitoring data. It charts the exposure pathway linking bathing sites to the outfalls in detail and is able to explore less frequent higher impact events than is logistically possible through current indicator sampling.

e. The current study also looked at the risks to other beach user populations using surfers as a conservative model.
7. References


direct detection and quantification of culturable and non-culturable *Escherichia coli* from agriculture watersheds. *Journal of Microbiological Methods* 69, 480-488


8. Appendices

Appendix 01 Newcastle Beachwater Quality and Baseline Reductions

Historical Newcastle Beachwatch monitoring data was provided by HWC and analysed with the following aims:

1. Document the extent to which the most vulnerable beaches achieved Category A and B water quality consistent with the Guidelines;
2. Determine how well contamination at the different beaches was correlated;
3. Document the extent to which high indicator level ‘events’ corresponded to Dry Weather or Wet Weather conditions;
4. Compare the observed Dry Weather water quality with that predicted by the different scenarios reported on in the earlier WRL study (Glamore et al., 2008) with a view to:
   a. Confirming that the model enterococci level estimates were generally in line with those actually seen and so providing validation/reality checking;
   b. Identifying which of the proposed indicator reduction scenarios (conservative v. slow inactivation) corresponded most closely with reality and hence which of the scenarios in the current assessment are most credible.
5. Estimate under Dry Weather conditions the magnitude of protection obtained for the beaches from WAS and Effluent by the current outfall arrangement with a view to estimating Baseline dilution + inactivation by the coastal zone waters (Appendix 14 Operational Application of Exceedence Probability Analysis To);

Summary Statistics for Water Quality (Beachwatch)

The Beachwatch program (Armstrong et al., 1997) has found that many of the spikes in indicator levels on Beaches can be ascribed to Stormwater after heavy rainfall when swimming is not recommended. Accordingly the data were filtered to remove those records which appeared associated with rainfall for the following reasons:

1. To determine water quality under dry weather conditions and confirm if they were high or not;
2. Compile a water quality data set whose statistics were conservatively likely to reflect only the impacts of the WWTP outfalls. Dudley Beach and Burwood Beach were particularly important for assessing discharge impacts because they are surrounded by a very high coverage of native vegetation rather than urban hard surfaces.

Table 1 shows the summary statistics calculated using the total data sets from two periods with and without the removal of rainfall impacted records covering:

1. all available data v. more recent data;
2. the total data set v. measurements from dry periods.

Table 1. Summary Statistics for Enterococci (cfu.100mL⁻¹) Beachwatch data collected for Newcastle Beaches in the Vicinity of Burwood Beach

<table>
<thead>
<tr>
<th>Period</th>
<th>Rainfall impact?</th>
<th>Statistic</th>
<th>Dudley beach</th>
<th>Burwood Beach South</th>
<th>Burwood Beach North</th>
<th>Merewether Beach</th>
<th>Bar Beach</th>
</tr>
</thead>
<tbody>
<tr>
<td>1996-2006</td>
<td>FALSE</td>
<td>No. of Measurements</td>
<td>377</td>
<td>378</td>
<td>378</td>
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<td>378</td>
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<tr>
<td>1996-2006</td>
<td>FALSE</td>
<td>Percentile 0.0 (a)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>1996-2006</td>
<td>FALSE</td>
<td>Percentile 0.1 (a)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
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<td>Statistic</td>
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<td>Burwood Beach North</td>
<td>Merewether Beach</td>
<td>Bar Beach</td>
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<td>--------------</td>
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<td>-0.7</td>
<td>-0.5</td>
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<td>336</td>
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</table>
Correspondence Between Contamination at Beaches

The 4 beaches all showed similar levels of contamination. However the estimated correlations were well below unity (Table 2). The best unsurprisingly were for the south v. north ends of Burwood Beach. But even with these correlations were only 0.75.

The next highest was the correlation between Merewether and Bar which lie in much the same direction from the discharge points. It was considered that conclusions about risks for Merewether Bather were probably applicable to Bar Beach.

Dudley Beach was least well correlated with the other beaches though the numbers of particles reaching the beaches in practice were similar.

A possible reason may be that contamination is likely to be very dependent on wind and current speed and direction.

Two way ANOVA was used to distinguish variance due to different sampling days with that arising from locations. It appeared that there were small but still significant differences between the beaches (Table 3).
Table 2. Correlation (R Coefficient) Between Beach Enterococci Level 2001-2006/no rainfall

<table>
<thead>
<tr>
<th>Data Set</th>
<th>Sampling Point</th>
<th>Dudley Beach</th>
<th>Burwood Beach sth</th>
<th>Burwood Beach nth</th>
<th>Merewether Beach</th>
<th>Bar Beach</th>
</tr>
</thead>
<tbody>
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<td>1.00</td>
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<td></td>
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</tr>
<tr>
<td></td>
<td>Merewether Beach</td>
<td>0.15</td>
<td>0.22</td>
<td>0.29</td>
<td>1.00</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Bar Beach</td>
<td>0.14</td>
<td>0.15</td>
<td>0.21</td>
<td>0.31</td>
<td>1.00</td>
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</tr>
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<td>0.36</td>
<td>0.39</td>
<td>0.59</td>
<td>1.00</td>
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</table>

Table 3. ANOVA (2 way without replication) comparing enterococci (log_{10}) measurements between Sampling Days and Sampling Stations

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<th>SS</th>
<th>df</th>
<th>MS</th>
<th>F</th>
<th>P-value</th>
<th>F crit</th>
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<td>0.934</td>
<td>5.296</td>
<td>5.77451E-71</td>
<td>1.186</td>
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<tr>
<td>Beaches</td>
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<td>4</td>
<td>0.692</td>
<td>3.924</td>
<td>0.004</td>
<td>2.382</td>
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<td>155.188</td>
<td>880</td>
<td>0.176</td>
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<tr>
<td>Total</td>
<td>363.439</td>
<td>1104</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Notes:
1. There was not statistically significant difference between the overall contamination levels seen at the different beaches when they were compared using 1 way ANOVA.

Evidence of Hazardous Events/Periods in Routine Indicator Monitoring Data

Examination of the historical data indicated that the median level of enterococci was below assay detection limits (1 enterococci cfu per 100 mL) and the water quality was generally consistent with Category A (95th percentile enterococci count < 40 /100mL).

Examination of the full historical data set however indicated there were numerous occasions when the enterococci and thermotolerant coliform levels were markedly elevated at one, or frequently, several beaches concurrently though the concurrency was not perfect (see above).

Some of these periods were associated with rainfall but the majority did not appear to be associated with high rainfall run off periods.

To quantify ‘non- rainfall’ v. rainfall derived (and presumably) events further the 2001-2006 enterococci data set classified into dry weather and wet weather and event and non-event using the following criteria:

- A ‘rainfall event’ was assumed to occur when there was:
  - >5 mm total rainfall on the day of bacteriological monitoring and the previous day;
  - or
  - 10 mm total on the day of measurement plus the previous 2 days
- A water quality event was defined as a day on which the level of enterococci was >90th percentile for that beach for the whole data set including the poorer years prior to 2001.
Including rainfall periods and poorer years tended to increase the 90th percentile and make this criterion more conservative. An event corresponded typically to an enterococci count > 10 cfu per 100mL which compares with a median enterococci level <1 cfu per 100mL.4

Table 4 shows that surprisingly the majority of ‘events’ could not be accounted for by rainfall. It was unclear whether these events reflected the influence of the waste discharge or alternatively contaminants transported in the Hunter River. However the latter would have been most evident following high rainfall periods.

This assumption is a conservative one necessarily. Other possible sources were the bathers themselves or occasional leaks from sewer mains. However two of the beaches are separated at a distance from urban areas. And the occurrence of ‘events’ concurrently at several beaches suggest the sources was not localised but rather but large and diffuse. The only source(s) that seems to fit this criterion are the outfalls. So provisionally it was concluded that the indicators seen during dry weather were the diluted residual numbers remaining from the ocean outfalls following dilution, dispersal away from the coast and inactivation.

Table 4. Water Quality ‘Events’ Characterised by High Enterococci Counts during Dry Weather Periods

<table>
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<th>Year</th>
<th>Rain</th>
<th>Dry</th>
</tr>
</thead>
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<tr>
<td>2001</td>
<td>14</td>
<td>47</td>
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<tr>
<td>2002</td>
<td>12</td>
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<td>2004</td>
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<td>43</td>
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<td>2005</td>
<td>19</td>
<td>42</td>
</tr>
<tr>
<td>2006</td>
<td>26</td>
<td>34</td>
</tr>
<tr>
<td>Average</td>
<td>19</td>
<td>37</td>
</tr>
<tr>
<td>Std Deviation</td>
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<td>10</td>
</tr>
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</table>

To quantify the residual contamination the ‘rainfall event’ data were next excised leaving dry weather event only. The enterococci counts associated with rainfall events were then graphed on an exceedence probability plot (Figure 1). This indicated increasing enterococci counts with decreasing likelihood.

4 The independent reviewer commented in regard to this ‘low’ event cut-off that:
   “The event definition of >10 enterococci/100ml is remarkably low suggesting very clean water in ‘global’ terms. This implies to me that rainfall flushing of animal derived faecal indicator organisms is not a major driver in these locations. Were there adjacent streams and surface water drains entering the bathing area? If not, this would be worth stating.”

We concur with this observation that it is low. But this is not surprising. The Beaches around Newcastle would be expected to be relatively clean by world standards for the following reasons:

1. The coastal zone is constantly swept by the East Australian Current - see Middleton, J. H., Cox, D. & Tate, P. (1996). The oceanography of the Sydney region. Marine Pollution Bulletin 33, 124-131;
2. Waters become deep (>100m) only a few km off-shore
3. Population density in Sydney region though high by Australian standards is low compared to most urban coastal areas having similar development levels. The only large urban area impacting the area is Newcastle (regional pop. ca 500,000)
4. There is no urban development north of Newcastle of comparable size.
5. The Hunter River which empties to the north of the city like most eastern Australian rivers is a small river by global standards, much of its dry weather flow is captured, agriculture is well upstream and is unlikely to impact quality during dry weather, and it empties into a large tidal estuary before moving out to sea;
6. The beaches especially the southern two have small proximate catchments which are largely forested;
7. The large majority of Newcastle is sewered (indeed BBWTP is the main treatment system);
8. The latitude is ca 33°S so sunlight is relatively strong and likely to be effective.

An indication of the water quality that can be expected is provided by Beachwatch data Armstrong, I., Higham, S., Hudson, G. & Colley, T. (1997). The Beachwatch pollution monitoring programme: Changing priorities to recognize changed circumstances. Marine Pollution Bulletin 33, 249-259. The Newcastle beaches are probably most comparable to Sydney’s ‘northern beaches’ where the median enterococci count was still ca 5cfu.100mL-1 compared to the current Newcastle median of ca 1 enterococcus.100mL-1.
Though the data did not have the same potential for supporting assessments of the impact of the low recurrence high impact events that modelling did (this would have required many more data measurements) it did still provide a data set against which model estimates could be compared or calibrated in part.

Notable features included:
1. The PDFs from the four reference beaches were virtually identical including Burwood Beach, which is closest to the outfalls supporting the contention that the contaminant source was not localised to one beach and Burwood is no worse because of its proximity.
2. The levels of enterococci exceeded the 100 per 100 mL around the 99.8th percentile (note the scale on the graph is per L for comparison with later plots).

![Figure 1 Exceedence Probability Plots of Enterococci Levels for Dry Weather Conditions at the Beaches from Historical Data](image)

Notes:
2. Enterococci counts are normally expressed as cfu per 100mL. In the present instance they are expressed per L because this is standard format we have adopted for all Exceedence Probability plots because pathogen levels are normally expressed per L.
3. n=220 so limiting percentile which can be estimated 0.995 (99.5%).
4. X axis is shown as 1-Percentile due to crowding of data on standard arithmetic plot.
5. Half of the values were below detection limit, there were treated as half detection limit values for calculation purposes.

**Predicted v. Observed Water Quality**
The previous WRL report (Glamore *et al.*, 2008) undertook modelling of the microbial levels in the Beach swimming zones prior to calculation of ‘Dry Weather’ water quality statistics. It was assumed that the discharge level of enterococci was $10^5$ per 100 mL$^{-1}$. The modelling outputs included predictions for median and 95th percentile enterococci levels in the bathing zone under decay ($T_{90}$s of 20 and 50h for <4 m and >4 m respectively) and conservative conditions.
In effect this work provided hypotheses on current water quality which:

1. could be compared to those observed in practice at each Beach and documented in this Appendix;
2. provided a reality check on the model itself.

Contaminant levels were dominated by the treated effluent. Allowing for the fact that the actual effluent discharge levels were slightly higher (see Appendix 11 Water Quality and Hydrological Strategic Monitoring and Project Implementation). Nevertheless comparison of the observed and predicted data (Table 4) showed the following:

1. The predicted and observed values were of comparable order of magnitude;
2. Allowing for the high levels of enterococci discharged the predicted 95th percentile enterococci counts should have been of the order of 50 to 100 \(\text{100mL}^{-1}\) under conservative conditions and ca 5 to 20 \(\text{100mL}^{-1}\) under decay conditions;
3. The observed dry weather water quality were virtually midway between these extremes;
4. The median observed levels were close to or below the detection limit consistent with the predicted data. The main caveat was that the predicted median dilution was >10^6 whereas the monitoring data was more consistent with a dilution closer to 10^5.

It was concluded that as far as it was possible to tell:

1. The model provided reasonable order of magnitude predictions of water quality under less dilute conditions;
2. Inactivation of effluent microorganisms in the coastal zone was likely to be slow and the inactivation rates used in the interpretation of the QMRA risk estimates needed to consider the conservative (dilution only) situation as well as the high inactivation rate scenarios;
3. The decision to omit dark inactivation from modelling was reasonable.

Table 4. Selected Summary Statistics for Enterococci (cfu.100mL^{-1}) Beachwatch data collected for Newcastle Beaches in the Vicinity of Burwood Beach

<table>
<thead>
<tr>
<th>Period</th>
<th>Statistic</th>
<th>Dudley beach</th>
<th>Burwood Beach South</th>
<th>Burwood Beach North</th>
<th>Merewether Beach</th>
<th>Bar Beach</th>
</tr>
</thead>
<tbody>
<tr>
<td>All Data including Rainfall Impacted days 2001-2006</td>
<td>Percentile 0.95 (a)</td>
<td>23</td>
<td>52</td>
<td>41</td>
<td>43</td>
<td>42</td>
</tr>
<tr>
<td></td>
<td>Percentile 0.95 (b)</td>
<td>20</td>
<td>30</td>
<td>30</td>
<td>33</td>
<td>35</td>
</tr>
<tr>
<td></td>
<td>Median (a)</td>
<td>0</td>
<td>1</td>
<td>1</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td>Data for days without Rainfall Impact 2001-2006</td>
<td>Percentile 0.95 (a)</td>
<td>16</td>
<td>16</td>
<td>16</td>
<td>24</td>
<td>21</td>
</tr>
<tr>
<td></td>
<td>Percentile 0.95 (b)</td>
<td>11</td>
<td>14</td>
<td>14</td>
<td>16</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>Percentile 0.95 (c)</td>
<td>22</td>
<td>19</td>
<td>22</td>
<td>20</td>
<td>24</td>
</tr>
<tr>
<td></td>
<td>Median (a)</td>
<td>0</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Median (c)</td>
<td>0.8</td>
<td>1.3</td>
<td>1.3</td>
<td>1.4</td>
<td>1.6</td>
</tr>
<tr>
<td>Conservative 2007</td>
<td>Percentile 0.95 (d)</td>
<td>24</td>
<td>25</td>
<td>25</td>
<td>41</td>
<td>30</td>
</tr>
<tr>
<td></td>
<td>Median (d)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>2007 Time Decay</td>
<td>Percentile 0.95 (d)</td>
<td>3</td>
<td>2</td>
<td>2</td>
<td>13</td>
<td>6</td>
</tr>
<tr>
<td></td>
<td>Median (d)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>

Notes:
1. Data marked:
   a. refers to statistics estimated on the primary data set using Excel functions such as Percentile().
   b. were calculated after log transforming the data. Below detection limit data were substituted with a half detection limit (i.e. 0.5 enterococci per 100 mL). Where indicated they are still in log_{10} transformed format. In the case of the 95th percentile this was estimated using the (arithmetic) average and standard deviation and using the Norminv() function to estimated the 95th percentile.
   c. were estimated using Palisade @Risk curve fitting applied to the positive detection data.
   d. were modelled estimates from draft WRL assessment(Glamore et al., 2008) Tables 16 and 17.
2. These were based on the assumption of $10^5$ enterococci $\cdot 100\text{mL}^{-1}$ in the discharged effluent. The observed level geometric was actually $10^{3.5}$, about twice that assumed. Further given the skew of log normally distributed data the arithmetic average would in fact be higher still.

**Treated Effluent and WAS Baselines**

The WRL model (Glamore et al., 2008) is constrained somewhat by not being able to estimate residual levels following high dilutions corresponding to percentiles < the 50th percentile. The result is that in the report median reductions are reported in Table 18 as ‘<10^6’. This is an outcome of the scale of the computational task possible with current workstations. For modelling purposes this is not a conceptually fatal limitation as such high dilutions imply low risk. However it posed a problem for the QMRA if we were to concurrently estimate not only the risks of infection under event conditions but also under the dominant ‘Baseline’ conditions in the manner detailed in Appendix 14 Operational Application of Exceedence Probability Analysis To Hazardous Event Characterization. One option was to assume zero pathogens present. This was computationally possible and not unreasonable due to the potential for the Eastern Australian Current to transport much contaminants out to sea. However it was still problematic as the modelling of currents in the area showed fate and transport is complex and more importantly the routine monitoring did indicate median levels of enterococci unlikely to have come from stormwater of $ca100\text{mL}^{-1}$ (Table 1).

The alternative identified was to assume the observed reduction in enterococci numbers typically seen in monitoring was a surrogate for the barrier effect of the coastal zone under normal dominant discharge conditions. The initial (log$_{10}$)level $100\text{mL}^{-1}$ of enterococci was $ca5.4\pm0.3$ in effluent during dry weather. When compared with the geometric mean level of enterococci at the beach (Table 5) this indicated a total decimal reduction factor of $ca5.12\pm0.02$. The overall working hypothesis underlying the Baseline reduction in enterococci and pathogens in the coastal zone was that:

$$\text{Pathogen/Enterococci level of seawater at the beaches} \approx \left(\text{Pathogen/Enterococci level in secondary effluent} + \text{Pathogen/Enterococci level of WAS } \times 0.1\right) / \left(\text{Reduction (factor) provided by the coastal zone}\right).$$

In defining this relationship we assumed:
1. (conservatively) that all the ‘Dry Weather’ enterococci came from the effluent and WAS discharges;
2. the typical water quality measured in the historical beach data assessment (Table 1, Figure 1 above) was a result solely of the reduction of the combined effluent and WAS discharges (see Figure 2);
3. the reduction factor for enterococci was a reasonable conservative approximation for the normal reductions in pathogens generally.

To assess how reasonable these assumptions were for estimating ‘Baseline’ we constructed exceedence probability charts for enterococci in treated effluent and WAS being reduced by factors of $10^4$, $10^5$, $10^6$ and $10^7$ and compared these to Figure 1. The closest correspondence to observed enterococci counts in Figure 1 and the relative flows was that secondary effluent was reduced by $10^5$, and WAS by $10^6$ (Figure 2). As a result these reduction factors were used as defaults within the QMRAs. They were also considered reasonable/conservative because:
1. The WRL modelling indicated median DRs > 6 for both effluent and WAS;
2. Event modelling showed the 95th percentile DRs for WAS was >5.
3. The comparison employed the conservative assumption that all non-rainfall associated enterococci came from the discharges.
These factors ($10^5$ and $10^6$) were also used in the QMRA to estimate the approximate conservative level of risk normally arising from the discharges and for comparison with the more precise reduction estimates derived from the hydraulic model.

Subsequently these reduction factors were estimated more precisely using the @Risk software in a trial and error fashion using enterococci level PDFs and discharge flows. The results are shown in Table 6. It can be seen that the order of magnitude values are very similar and the $10^5$ and $10^6$ are slightly on the conservative side.

<table>
<thead>
<tr>
<th>Statistic</th>
<th>Bar Beach</th>
<th>Merewether Beach</th>
<th>Burwood Beach</th>
<th>Dudley Beach</th>
</tr>
</thead>
<tbody>
<tr>
<td>Arithmetic Average</td>
<td>4.6</td>
<td>5.0</td>
<td>4.1</td>
<td>4.4</td>
</tr>
<tr>
<td>Geometric Mean</td>
<td>1.8</td>
<td>1.7</td>
<td>1.6</td>
<td>1.6</td>
</tr>
<tr>
<td>95th Percentile</td>
<td>21</td>
<td>24</td>
<td>16</td>
<td>16</td>
</tr>
<tr>
<td>99th Percentile</td>
<td>46.8</td>
<td>48.2</td>
<td>46.4</td>
<td>68</td>
</tr>
<tr>
<td>Maximum</td>
<td>120</td>
<td>171</td>
<td>128</td>
<td>120</td>
</tr>
<tr>
<td>$\log_{10}$ average</td>
<td>0.261</td>
<td>0.230</td>
<td>0.213</td>
<td>0.206</td>
</tr>
<tr>
<td>$\log_{10}$ Std Dev</td>
<td>0.583</td>
<td>0.585</td>
<td>0.568</td>
<td>0.570</td>
</tr>
<tr>
<td>Count</td>
<td>221</td>
<td>221</td>
<td>221</td>
<td>221</td>
</tr>
<tr>
<td>Decimal Reduction</td>
<td>5.09</td>
<td>5.12</td>
<td>5.14</td>
<td>5.15</td>
</tr>
</tbody>
</table>

Table 6. Average ($\log_{10}$) Baseline Decimal Reductions Estimated for Each Beach and Discharge

<table>
<thead>
<tr>
<th>Beach</th>
<th>Effluent</th>
<th>WAS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bar</td>
<td>5.25</td>
<td>6.03</td>
</tr>
<tr>
<td>Merewether</td>
<td>5.28</td>
<td>6.06</td>
</tr>
<tr>
<td>Burwood</td>
<td>5.30</td>
<td>6.08</td>
</tr>
<tr>
<td>Dudley</td>
<td>5.31</td>
<td>6.09</td>
</tr>
</tbody>
</table>

![Graph of enterococci and dilution](enterococci_2nd Nominal Dilution 5_Shoreline bathers_.png)
b. Figure 2. Exceedence Probability Plots Modelled for Enterococci in Seawater Reduced by $10^5$ (secondary effluent) and $10^6$ (WAS)

Notes:
1. a. is for Secondary Treated Effluent, b. is for WAS
Appendix 02 Models for Microbial Risk Assessment in the Natural Environment in the Literature

The implication in the guidelines is that hydrodynamic/hydraulic and QMRA modelling should be combined. This is in line with feedback from NSW Health and DECC and the Community Reference Group when the current project approach was formulated.

As far as we have been able to determine from the literature such a combination of full QMRA and outfall discharge transport modelling has never been fully undertaken. However, there is increasing use of a combination of hydrodynamic modelling with indicator behaviour (Falconer et al., 2001; Harris et al., 2004; Kashefipour et al., 2002; Kay et al., 2005; Lin et al., 2008). Such modelling focuses on understanding the generic movement of microorganisms overall rather than estimating the quantitative risk posed by infrequent but high impact Hazardous Events. The work of Harris et al. illustrates the approach (Harris et al., 2004) and is essentially the same as that already applied in the case of Burwood Beach previously (Glamore et al., 2008).

Further there are many analogous examples where QMRA and environmental transport modelling have been combined with a view to protecting human health. A study that did directly combine data on protozoan pathogens with hydraulics was one in Netherlands river waters which combined inputs levels and dispersion modelling (Medema & Schijven, 2001) to determine the relative contribution of WWTPs to total loads seen in rivers. Other Dutch work has combined hydrology and QMRA with a focus on groundwater (e.g. Schijven et al., 2005; Schijven et al., 2006).

At WRC we have also undertaken a range of projects which combine QMRA with environmental modelling to estimate risks to human populations from contaminated vegetables, groundwater and surface waters (Charles et al., 2004; Charles et al., 2008a; Charles et al., 2008b; Petterson et al., 2001a; 2002; Petterson et al., 2001b; Signor et al., 2008; Signor et al., 2007; Signor et al., 2005).

As a result of discussions with Hunter Water and other stakeholders it was decided in principle to move beyond the characterisation of indicator behaviour to QMRA involving risk estimation. Normally such a project would have required 2-3 years of surveys, stakeholder negotiations and system analysis before undertaking a fully hydro-dynamically integrated QMRA. However, in the case of Burwood Beach:

- Three integrated hydraulic models were already available which covered the summer months off Newcastle’s beaches and appeared possible to reconfigure to suit a conventional QMRA analysis;
- Additional data existed on wind and current speed from an earlier project which could be used to simulate pathogen fate and transport during winter conditions;
- Computer speeds have increased to a point where a sufficiently large number of simulations could be undertaken in a more realistic period than was previously possible;
- A QMRA metamodel had been previously developed for the EU MicroRisk project (Medema et al., 2006; Petterson et al., 2006; Roser et al., 2006b) for rapidly undertaking large numbers of different QMRA simulations and had proved possible to adapt to generate modelling needs in line with NSW Health requirements (Roser et al., 2006a).

This left only the characterisation of pathogen content of the source material and agreement among stakeholders on which pathogens, human populations and locations to assess and an agreement on a credible consumption level for surfers (200mL per bathing occasion).
Appendix 03 Justification for Project Design

The critical concern to the CRG, NSW Health and DECC, could be summarised as whether under rarer occasions high quantities of pathogens from treated sewage and WAS reached the nearby bathing zones in loads and at frequencies likely to pose a higher risk to bathers than was indicated by the general suitability quality assessed using routine indicator monitoring.

These concerns appear to be in line with the natural water recreation guidelines (NH&MRC, 2008). Guideline extracts relevant to the Burwood Beach situation and supporting this contention are presented below which support the proposed assessment strategy (Roser & Stuetz, 2008a) being appropriate:

“Exceptional circumstances are known periods of higher risk such as during an outbreak involving a human or other pathogen that may be waterborne (e.g. avian botulism — where outbreaks of avian botulism occur, swimming or other aquatic recreational activities should not be permitted), or the rupture of a sewer in a recreational water catchment area etc. Under such circumstances the classification matrix may not fairly represent risk/safety.”

(Section 5.3)

“(As part of management) produce and verify flow charts for faecal pollution from sources to recreational exposure areas. This may require a new sanitary inspection. The series of flow charts should illustrate what happens to feeder waters before they join the recreational water body in sufficient detail for potential entry points of different sources of faecal contaminants to be pinpointed and any detected contamination to be traced.”

(Table 5.6)

“Quantitative microbial risk assessment can be used to estimate indirectly the risk to human health by predicting infection or illness rates for given densities of particular pathogens, assumed rates of ingestion and appropriate dose–response models for the exposed population. Application of this process to recreational water use is constrained by the current lack of specific water-quality data for many pathogens and the fact that pathogen numbers (as opposed to faecal indicator organism numbers) vary according to the prevalence of specific pathogens in the contributing population and may exhibit seasonal trends.”

(Section 5.3)

“Although outfall length is relevant, it is usually less important than proper location and effective diffusion. An effective outfall is one that is properly designed with sufficient length and depth of diffuser discharge to ensure that sewage is unlikely to reach the recreational area. In public health terms, it is generally assumed that dispersion, dilution, sedimentation and inactivation (through sunlight, predation, natural die off etc) after discharge from a piped outfall will lead to a certain degree of safety. In practice, a number
of factors reduce the efficiency of these processes, with the most important being factors that lead to the rapid movement of sewage into recreational areas. For example, where sewage is warmer and less saline than the receiving water, it may mix poorly and form a floating slick that will be easily influenced by wind and may therefore severely pollute even distant recreational areas. Properly designed and operated diffusers on the outfall should prevent the formation of such slicks. Also, it is possible to reduce the risk from floating slicks by recognising periods of high risk (e.g. during on-shore winds) and taking appropriate action, such posting advisory notices or zoning or banning water contact activities. Coastal currents and tides may give rise to similar problems and may be recognised and dealt with the same way.”

(Section 5.4)

“Because contamination may be triggered by specific and predictable conditions (eg rainfall run-off), local management actions can reduce or prevent exposure at such times. Provided such actions can be shown to be effective, the recreational water environment classification may be upgraded to a more favourable level. However, a reclassification should initially be provisional and time limited. It may be confirmed if the efficacy of management interventions (e.g. advisories) is verified during the following bathing season, otherwise it will automatically revert to its original classification.

Some events triggering water contamination can be measured by simple means, such as rainfall gauges, detectors on stormwater overflows and the like. Others may require more sophisticated approaches, such as modelling. Two main approaches have been used for real time prediction of faecal indicator organism levels at recreational area compliance points.

• One method is to use background conditions to calibrate a statistical model, typically based on the relationships of multiple predictor variables, such as:
  o preceding rainfall;
  o wind direction;
  o tides and currents;
  o visible or modelled plume location;
  o solar irradiance (and turbidity of water); and
  o physicochemical parameters of water quality.

• The alternative approach is to construct a nearshore hydrodynamic model linked to a water-quality model predicting levels of faecal indicator organisms (Falconer et al. 1998).”

(Section 5.5.4)
Appendix 04 Risk Management and HACCP

The NHMRC (2008) Guidelines make it clear that any monitoring should be linked to management activities especially those identified by the HACCP process. What these are and how the current project appears to fit with them are shown in Table 1.

Table 1. Addressing of HACCP Principles

<table>
<thead>
<tr>
<th>Principle/Major Action</th>
<th>What this means in Practice</th>
<th>How this has been is to be Addressed</th>
</tr>
</thead>
<tbody>
<tr>
<td>Assemble the team</td>
<td>Form a team to steer the overall process. Composition of the team should represent all stakeholders and (as much as possible) cover all fields of expertise. Consider representatives of health agencies, user groups, the tourism industry, the water and sewerage industry, communities, relevant authorities (e.g. resource management, environment), potential polluters, and experts in hazard and risk analysis and other fields.</td>
<td>The project has clearly engaged regulators, specialist consultants and the communities and has done this early in the process.</td>
</tr>
<tr>
<td>Collate historical information</td>
<td>Summarise previous data from sanitary inspections, compliance testing, utility maps of sewerage, water and stormwater pipes and overflows. Determine animal faecal sources for each recreational water body. Check development applications and appropriate legal requirements. If no historical data are available, collect basic data to fill the data gap or deficiency.</td>
<td>Historical indicator data has been collected along with details of rainfall and bypass conditions. Historical data on hydraulic loadings at the plant Collection of scientific papers on wastewater discharge Data initially analysed in CH2M Hill (2008) study and the WRL/WRC Study (e.g. Glamore et al. 2008)</td>
</tr>
<tr>
<td>Produce and verify Flow charts</td>
<td>Produce and verify flow charts for faecal pollution from sources to recreational exposure areas. This may require a new sanitary inspection. The series of flow charts should illustrate what happens to feeder waters before they join the recreational water body in sufficient detail for potential entry points of different sources of faecal contaminants to be pinpointed and any detected contamination to be traced.</td>
<td>Exposure pathways constructed consistent with geographic information available in previous reports. Add information on ocean currents and WRL/WRC hydrodynamic modelling so as to attach numbers to the various barriers and paths shown in the current chart. Illustrative flow paths within the WWTP identified</td>
</tr>
<tr>
<td>Hazard analysis</td>
<td>Identify human versus different types of animal faecal pollution sources and potential points of entry into recreational waters. Determine the significance of possible exposure risks (based on judgment and on quantitative and qualitative risk assessment, as appropriate). Identify preventive measures (control points) for all significant risks.</td>
<td>Potential hazards identified by definition to be material released by BBWWTP finding their way to the beaches and surfing areas. Focus of this study is pathogens. Initial Screening risk assessment was undertaken. This study extends this initial work.</td>
</tr>
<tr>
<td>Control points</td>
<td>Identify those points or locations at which management actions can be applied to reduce the presence of, or exposure to, hazards. Examples include signage at</td>
<td>Provisionally the key ones seem to be:</td>
</tr>
<tr>
<td>Principle/Major Action</td>
<td>What this means in Practice</td>
<td>How this has been is to be Addressed</td>
</tr>
<tr>
<td>------------------------</td>
<td>-----------------------------</td>
<td>-------------------------------------</td>
</tr>
</tbody>
</table>
| beaches, municipal sewage discharge points, treatment works operations, stormwater overflows and illegal connections to drains. | - Treatment process  
- Bypass point  
- Plant exits  
- Outfall diffusers  
- Thermocline  
- Solar radiation  
- Wind and current off shore or along coast  
- Recreation points | |
| Critical limits | Determine measurable control parameters and their critical limits. Ideally, assign target and action limits to pick up trends towards critical limits (e.g. > 10 mm rainfall in previous 24 hours or notification of sewer overflow by local agency). | A benchmark probability of 0.01 of Gastroenteritis was proposed by NSW Health |
| Monitoring | Establish a monitoring regime to give early warning of exceedences beyond critical limits. Those responsible for the monitoring should be closely involved in developing monitoring and response procedures and protocols. Note that monitoring is not limited to water sampling and analysis but could also include, for example, visual inspection of water users and potential sources of contamination or flow and overflow gauges. | Beachwatch is already in place and will provide long term exceedence data for enterococci.  
This project is design to implement strategic monitoring. |
| Management actions | Prepare and test actions to reduce or prevent exposure in the event of critical limits being exceeded. Examples include building an appropriate treatment and/or disposal system, training personnel, developing an early warning system, issuing a media release and (ultimately) closing the area for recreational use. | Current strategic modelling and monitoring is designed to understand the system better with management in mind.  
Repair of diffusers is proposed.  
Upgrades of BBWWTP are proposed contingent on the strategic monitoring findings. |
| Validation/verification | Obtain objective evidence that the envisaged management actions will ensure that the desired water quality will be obtained or that human recreational exposures will be avoided. This step would draw on the literature and in-house validation exercises.  
Obtain objective data from auditing management actions that the desired water quality or change in human exposure is in fact obtained and that good operational practices, monitoring and management actions are being complied with at all times. | Assessment of dilution based on of dye tracing has been done by WRL.  
WRL model predictions for enterococci are being compared with those actually observed.  
All new data includes QA/QC.  
Validation/verification is incorporated in the Uncertainty analysis. |
| Record keeping | Ensure that monitoring records are retained in a format that permits external audit and compilation of annual statistics. These should be designed in close liaison with those using the documents and records. | Development of information repository and database covering the monitoring of the WWTP and the outputs of the hydraulic modelling. |
Appendix 05 Final Study Approach

Hydraulic and QMRA Modelling proved relatively straightforward and appears to be very informative in respect to Hazardous Event risks and their causes and impacts.

The overall set of tasks proposed and in the end implemented is summarised in Figure 1. The different process symbols and boxes represent a division as follows:

1. Actual input measurement data which is being obtained either by local experimental work or from the literature.
2. Processes which act on and generally reduce the levels or loads of contaminants.
3. Hydrodynamic model components which simulate for the coastal zone of interest the transport and inactivation of pathogens and enterococci.
4. Probability density functions describing:
   a. The initial contaminant levels;
   b. Contaminant levels after treatment and partitioning;
   c. Contaminant levels to which bathers would be exposed;
   d. Contaminant Doses;
   e. Estimated risk of illness or infection;
5. Support and quality control and assurance activities.

Figure 1 is structured as follows:

1. Different data input types are shown in color coded boxes (key is a).
2. Primary measurements being collected de novo are shown in light blue square boxes while ancillary information from the literature of derived conceptually is shown in violet hexagons.
3. The elements of the exposure pathways are shown in green (b to c). The main path itself is indicated by the heavy black arrows.
4. The key modifying process or modifying factors are shown in yellow octagons.
5. The location of the hydrodynamic model in shown in red ovals.
6. The principal outputs of the model are shown in d. QMRA does not generate a single value but rather an array which is in effect a probability density function or PDF. Because of the nature of the hydrodynamic modelling the output arrays are also in effect a timeseries.

As noted above the scale of modelling is constrained somewhat by the issue of combinatorial explosion i.e. in mathematics the effect of functions that grow very rapidly as a result of combinatorial considerations (see Appendix 12 Guidelines, Combinatorial Explosion and the Scale of Risk Modelling). Modelling generates not one risk estimate but many combinations of input and the barrier process sub-models (= functions) which might be run depending on the exposure pathway scenarios selected. To address this primary scenarios considered likely to yield the most useful information will be simulated first. Remaining questions will be dealt with by inference from these model outputs or if it is essential running further simulations on a needs basis.

In addition to generating input for risk modelling the hydraulic modelling process generated raw particle transport data which should give additional insights into the behaviour of pathogen levels in the coastal zone more generally and may aid the decision support and beach management more generally (e).

The data collected here is likely to have a range of future uses relating to HWC’s core business additional to the primary study here. For example the measurements of pathogens and indicators in
raw sewage should provide an excellent data set supporting better understanding of the impacts of sewage overflows and stormwater in Newcastle and indeed in Australia generally. Detailed systematically collected data sets of this type are known to not be very abundant yet Australia wide.

In both quantitative and qualitative risk assessment it is desirable to collect data which will inform on the uncertainty of the assumptions, inputs, barriers and outputs. To some extent uncertainty is accounted for by the use of probabilistic modelling. But pathogen and microbial assay technology has its own uncertainties separate to the variability in the primary input data. To this end a range of QA/QC measurements are also being collected. The risk assessment includes a Chapter on Uncertainty.

In addition to undertaking standard QA/QC a WAS microcosm system will be studied to determine whether the rates at which microorganisms are inactivated in diluted WAS are comparable to rates in seawater.

Figure 1. Relationship between data and other information collected, modelling and outputs.
Appendix 06 Other Strategic Elements

Systematic Issue Identification
From discussions with NSW government representative and the community reference group the primary issues were the water borne pathogen risk arising from:

1. WAS generally;
2. WAS following modifications to Burwood WWTP required to cope with future population increases leading to an increase in the total effluent and WAS discharged;
3. Pathogens from secondary effluent and WAS reaching the beach during rare periods when reduced dilution, surfacing of effluent and/or on-shore transport occurred;
4. Whether surfers were substantially more at risk.

It was recognised that the water quality during normal median periods was very high and that the 95th percentile enterococci counts at all beaches achieved was probably Microbial Category A (NH&MRC, 2008) which is notable for being markedly more stringent than primary recreational quality targets in the previous guidelines (National Health and Medical Research Council, 1990).

Construct Exposure Pathways
The exposure pathway is the coastal surface waters off these beaches where mixing and dispersion of the effluent and WAS takes place.

Estimate Exposure Levels
Baseline levels have been assumed to arise from a high degree of dilution/inactivation of effluent and WAS consistent with the indicator levels of the material discharge, the levels of indicators seen in the beach zones during dry weather, and the fact that the volume of effluent is ca 10 times that of the discharged WAS.

Exposure levels during Hazardous Event conditions have been estimated by integrating (Monte Carlo modelling) wastewater PDFs with the 3 component ocean hydraulic model. The model generates 15 minute timestep reduction/inactivation/dilution estimates reflecting preceding wind, current and tidal conditions. Inactivation is assumed to be driven by solar radiation to different degrees or to not occur at all in which case removal is due primarily to dilution and dispersion.

Baseline exposure levels have been assessed at being ca $10^{-6}$ of the original WAS level and $10^{-5}$ of the original secondary effluent level.

Reduction/Dilution/Inactivation reduction factors have been assessed as ranging from $10^2$ to $10^{10}$ [Decimal Reduction (DR) = 2 to 10] depending on the timestep and the exposure scenario.

Overall Strategy
For each risk scenario a Monte Carlo model has been constructed and used to generate a Probability Density Function (PDF) of estimated risk. In each case we have constructed an exceedence probability plot for:

1. Pathogen level;
2. Risk of infection from that pathogen;
3. Risk of illness from that pathogen.

These plots cover ‘Baseline’ conditions and Hazardous Event conditions.

We have also:
1. Calculated a range of statistics on the behaviour of the coastal waters in respect to their expect reduction of pathogen particle numbers;
2. Plotted the behaviour of the on-shore particle movement in respect to time to assess what patterns there might be relevant to future monitoring and management;
3. Calculated the following statistics which have been extracted into a database for auditing/checking purposes:
   d. Average (aggregate) risk of illness or infection (incorporates low and high values together);
   e. Upper 95th percentile risk of illness or infection;
   f. Median 85th, 90th, 99th percentiles.

The risk estimates are in would then be compared with the risk benchmarks. The key benchmark proposed against which calculated risks is the 1% overall GI illness risk.

The option also exists for comparing these to a 0.1% individual pathogen infection/illness risk noting this is not a standard benchmark for an individual pathogen.

As indicated above QMRA based risk characterisation has involved:
   1. Construction of multiple probabilistic barrier models corresponding to different indicative exposure pathways and a range of representative scenarios reflecting potential risk situations;
   2. Calculation of risk posed by several model hazards or hazard indices (enterococci);
   3. Comparison against ‘Benchmarks’.

The exposure assessment approach in general has been outlined in the introduction to this part (D) and presented in the previous planning reports (Roser & Stuetz, 2008a). The Target Populations are healthy adult swimmers in the surf zone & surfers 200 m immediately off the surf zone location. The target locations proposed are Bar Beach, Merewether Baths, Burwood Beach & Dudley Beach. Inactivation has been modelled using/based on:
   1. conservative “dark inactivation” rate constants from the literature developed for seawater, or in their absence freshwater;
   2. cumulative inactivation reflecting the cumulative dose of solar radiation to which particles would be exposed during their travel from the outfalls to the points of exposure (Rayner et al., 2009).

Hydraulic modelling has generated data arrays:
   1. Corresponding to WAS and Treated effluent particles and simulating their fate and transport over 2 days (conservative particles) and 7 days (particles with S90S of 3, 15 or 75 MJ.m⁻²)
   2. Using a three component hydraulic model estimating particle dilution, inactivation (loss of mass), travel time based on time spent at different depths (or subject to feasibility), cumulative solar radiation dose exposure and movement outside of the coastal areas;
   3. That covers summer and winter, and the 8 exposure locations.

Ground-truthing/validation/reality checks proposed were:
   1. A targeted literature review of input data to verify new data are was consistent with observations in the literature;
   2. Comparison of particle levels remaining under the different inactivation scenarios;
   3. Comparison of predicted with actual enterococci numbers in the surf zone;
   4. Measurement of solar radiation penetration into seawater;
   5. Assessment of the behaviour of microbial indicators during the treatment process.
Appendix 07 Scope of Work Planned at Project Commencement

Strategic Monitoring and Risk Modelling
A strategic monitoring program was developed together with risk modelling and data analysis (Roser & Stuetz, 2008b; Roser et al., 2008). As a result the following work was proposed and undertaken:

1. Characterise Infection and Illness risk using QMRA scenario modelling which:
   a. Uses local estimates of against agreed benchmarks;
   b. Models risk scenarios covering:
      i. between population differences;
      ii. geographic variation;
      iii. 2007 v. 2030 predicted discharges;
      iv. Summer v. winter mixing conditions;
      v. A range of inactivation rates;
      vi. Between pathogen variation;
2. Integrate QMRA and pathogen fate and transport modelling;
3. Undertake in support of this modelling:
   a. Strategic water quality monitoring data and short term experiments aimed at filling data gaps and checking;
   b. A review of available oceanographic data and whether it is sufficiently representative of conditions off Burwood Beach;
4. Use literature relevant to these issues:
   a. as a reality check;
   b. as an aid to interpretation;
   c. to identify defensible/efficient modelling scenarios;
   d. to identify appropriate ‘sensitivity test’ values;
   e. to fill data gaps where it was impractical to collect local data (e.g. seawater ingestion, dose response relationships);
5. Develop methods for efficiently presenting and communicating the risk estimates.

Analysis of Historical Enterococci Data
A reanalysis of historical enterococci data was seen as providing:

1. a check of whether the beaches were typically Category A;
2. data which could be used to estimate how much reduction in microbial level was in practice achieved by the outfall discharge waste disposal system under normal ‘Baseline’ conditions;
3. an indication of contamination levels at percentiles higher than the 95th where evidence of rare/Hazardous Events might be encountered.

This work is detailed further in Appendix 01 Newcastle Beachwater Quality

Locality Specific pathogen level and load data
A survey of BBWWTP wastewater and WAS quality was designed which could be undertaken by local contract water quality laboratories as well as HWC. It was aimed at providing sufficient data to estimate probability density function (PDF) coefficients defining the typical Baseline quality of the raw screened sewage, secondary treated effluent which could be used subsequently in QMRA modelling.
The index pathogens proposed for this analysis were *Cryptosporidium* spp., *Campylobacter* spp., and Rotavirus. It was also proposed to measure enterococci and use the enterococci:infection correlation based algorithm which the NH&MRC(2008) guideline microbial categories are based on, to estimate total gastrointestinal illness probability. [Subsequently measurements of *Giardia lamblia* and Adenovirus levels were also undertaken for reasons explained below].

To assess the extent to which pathogen levels were likely to be representative, intensive indicator monitoring was also proposed to assess variation diurnally and over the course of the survey.

Finally it was proposed to measure waste stream flows to determine:
1. Whether the measurements collected reflected normal dry weather flows or were impacted by wet weather surges;
2. The reduction in pathogen numbers achieved by the secondary treatment processes;
3. The extent to which pathogens were concentrated in the WAS and partitioned between the two streams.

A limitation of the pathogen survey study was that gastrointestinal disease prevalence can often be seasonal (e.g. Institution Gastroenteritis Table 3 Communicable Diseases Branch, 2008) and would unlikely to pickup peaks associated with outbreaks (e.g. Lee *et al.*, 2001).

As data on such seasonality was likely to be rare the approach for incorporating the approach was seen to be via sensitivity testing.

**WAS Characteristics**

Whether the WAS posed special pathogen related risk was to be addressed by:
1. Separate pathogen analysis, hydraulic modelling and QMRA of WAS and secondary effluent streams;
2. Modelling of 2007 and 2030 discharge impacts;
3. Experiment based measurements of indicator inactivation.

**Pathogen survival**

The issue of variable pathogen survival was proposed to be addressed by:
1. Generating inactivation rate data covering rates ranging from the most rapid reasonably conceivable to the most conservative (dilution only);
2. The experiments on indicator and inactivation to determine at what rates this was likely to occur and how the rates varied between WAS and secondary effluent (*Appendix 16 Inactivation Studies Covering Microcosms and Water Transmissivity*).

One uncertainty identified for further strategic monitoring was ocean water transmissivity which would likely influence solar radiation driven inactivation.

**Population Risk Variation**

**Surfer v. Bather Risk**

Shoreline bather exposure was seen as the Baseline condition. Surfers were seen as being of additional/higher risk because:
1. They tend to be in the water longer;
2. The vigorous nature of surfing means they are likely to consume larger than average quantities of seawater;
3. Surfing occurs out to sea potentially closer to surfacing contaminants;
4. Surfers commonly are exposed during winter months.
Risk assessment needed to account for these differences.

**Others**
Beach users include different populations of users. To allow for this it was decided to:
1. Select relatively conservative dose response relationships;
2. Include various sensitivity tests in the scenarios tested.

**Seasonal Variation**
To allow for seasonal variation:
1. A set of hydrodynamic data on wind and current behaviour collected for the area during the winter of 1997 was assessed for inclusion in scenario testing;
2. This data set was to be used to generate ‘winter’ risk estimates;
3. Inactivation scenarios included the conservative ‘dilution without inactivation’.

**Fate and Transport of Pathogen in Coastal Waters**
The earlier hydraulic modelling undertaken by WRL was seen as the ideal platform for quantifying pathogen fate and transport in a format usable in QMRA.

**Integration of Hydraulic Modelling and QMRA**
There appeared to be two options available for integrating hydraulic modelling with QMRA:
1. Firstly to expand the hydraulic modelling with QMRA front and back ends;
2. Use the hydraulic modelling to generate generic particle dilution and mass ‘inactivation’ data for constructing PDFs.

The feasibility of each option was investigated.
Appendix 08 Hazardous Events associated with Coastal Zone Outfalls

Hazardous Events

Relationship to Beachwatch Data
The four beaches of interest are normally of a high quality based on the older NHMRC guidelines (National Health and Medical Research Council, 1990; NSW Department et al., 2008). Though these guidelines do not consider ‘Hazardous Events’ NSW Beachwatch does recognise this the potential in particular for stormwater (Armstrong et al., 1997). Though of more limited scope, Beachwatch (although it does not provide pathogen data for example) is still very useful by virtue of the length of the timeseries and its inclusion of enterococci as an analyte. Complementary to these records HWC provided concurrent daily rainfall data which could be used to excise periods with elevated indicator count arising most likely due to stormwater discharge. This made possible the development of a dry weather water quality data set where microbial contaminants could be expect to come (conservatively) from the outfalls. This was seen as providing a useful reality check on the risk assessment.

From an analysis of Beachwatch enterococci data and further to discussion in Section 1.4.2, under typical conditions (> 95% of the time) the water quality of the four Newcastle beaches corresponded to Microbial Category A and is therefore ‘Very Good’ as most of the time discharges from Burwood WWTP are transported out to sea away from the beaches (Glamore et al., 2008). However, there are several ways in which health risks to bathers could be higher at times than this rating indicates. Several are identified here and in the Guidelines and were discussed in consultations with DECC, NSW Health and the CRG:

1. Though Beachwatch sampling frequency is high it may be insufficient for fully characterising low frequency high impact health risk ‘events’ of concern notably periods of strong on-shore winds /currents leading to short travel times, limited dilution and pathogen inactivation, as suggested by earlier hydraulic modelling by Glamore et al., (2008), which is summarised in Section 1.5.9.2.;
2. The method used to estimate the 95th percentile could yield an incorrect value if not calculated appropriately (NH&MRC, 2008 p. 70);
3. There might be higher loads of viruses and other pathogens in the effluent discharges than indicated by enterococci concentrations e.g. during an outbreak or due to seasonal variation;
4. Pathogens might survive longer than indicators, especially viruses [compare coliform and viral inactivation rates in Table 5.8 (NH&MRC, 2008) with those summarised by USEPA (USEPA, 2001 ) and used by WRL in earlier modelling (Glamore et al., 2008)];
5. The WAS might have posed especially high risks;
6. Between population dose response variation - In particular surfers might be exposed to a higher than median risk due to their greater exposure to the aquatic environment;
7. In the future, increased discharge loads may increase risk.

This risk assessment was designed to address these issues and some others arising such as analytical resource constraints. This said the risk assessment does not invalidate the overall assessment undertaken by Beachwatch. Rather it complements the information available through this program.

Hazardous Events
The NH&MRC (2008, Table 5.13) Guidelines provide a scheme for assessing the overall microbial suitability of a recreation site.
The current study, however, was concerned overall with the issue of what are termed ‘Exceptional Circumstances’ or ‘Hazardous Events’. In the Guidelines a Hazardous Event is defined as:

“An incident or situation that can lead to the presence of a hazard (what can happen and how).”

Where a ‘Hazardous Event’ is seen as a possibility, further questions follow:

• What does a ‘Hazardous Event’ mean operationally?
• How are the ‘Hazardous Event’ conditions distinguished and can it be used in management and risk characterisation?
• What are the benchmarks that might be used to judge its severity?
• How should they be responded to?

The following extracts from the NH&MRC (2008) Guidelines indicate what ‘Hazardous Event’ means operationally in the case of recreational water pathogens:

1. Hazardous Events are seen as any situation where there may be:

   “Ingestion or inhalation of, or contact with, pathogenic bacteria, viruses and parasites, which may be present in water through contaminated discharges from run-off or faecal contamination from people or animals using the water, or may be present naturally”

2. The NH&MRC (2008) Table 5.13 includes in its criteria, for both sanitary assessment and microbial quality, ‘Exceptional Circumstances” which applies and appears to be synonymous with some types of Hazardous Event including the issue of interest here of inshore transport of microbial contaminants:

   “Exceptional circumstances are known periods of higher risk such as during an outbreak involving a human or other pathogen that may be waterborne.”

3. Finally the sanitary survey classification for ‘effective outfalls’ in NH&MRC(2008):

   “assumes that …climatic and oceanic extreme conditions are considered in the design objective (i.e. no sewage in the beach zone)”

By their nature, Hazardous Events are diverse in nature. But the information on them is also very limited and time consuming to gather (e.g. Nilsson et al., 2007). In the absence of full monitoring data on Hazardous Events or circumstances, the best available approaches to characterising them appear to be:

• What-if Scenario risk modelling of the kind we have undertaken previous (Khan et al., 2007b; Petterson et al., 2006; Roser et al., 2006a; Roser et al., 2006b);
• Targeted experiments, measurements and surveys to enhance the reliability of modelling assumptions;
• Use of high quality literature data for comparison, support and reality checking.

The concepts of ‘Hazardous Events’ and circumstances can also encompass other hazards too such as data reliability.

---

The principles behind Hazardous Event assessment are not new to water engineering. They are central to much hydrological management practice (Pilgrim & Doran, 1997) and are what was done in the earlier studies of Burwood Beach (Glamore et al., 2008).

What appears to be novel in the current study is the focus on pathogens and health risk combined with the level of detailed analysis proposed. To characterise ‘Hazardous Event’ risks a combination of risk modelling and complementary strategic monitoring was planned.

The Hazardous Event of most concern is illustrated in Figure 1. Most of the time the discharge diffusers are designed to inject and mix the treated effluent and WAS plumes into the coastal waters below the surface in the hypolimnion. On occasion though destratification can occur and plumes or a portion of the material may surface. Even then, most of the material is transported out to sea (a.). However, on occasion diluted waste plumes may be transported into the bathing zones (b.) as illustrated by these model contour plots (Glamore et al., 2008) noting the following:

1. The plots shown are not of actual quality. Rather they are model outputs reflecting wind, current water temperature etc. and a given emission Scenario;
2. The plots are shown not illustrate event form. They are not representative of event frequency;
3. At such times when plumes reach the beach there is still an extensive protective effect. For example in plot b. it can be seen that the thermotolerant coliforms have been reduced to \( \text{ca} \ 10^2 \) 100mL\(^{-1}\) compared to the assumed discharge level of \( 10^7 \) 100mL\(^{-1}\).
Figure 1. Illustrative Examples of Modelled Dispersion and Reduction of Thermotolerant Coliform numbers in Surfacing Effluent and WAS Plumes (Glamore et al., 2008)

Notes:
1. The initial assumed level in the effluent and WAS was $10^7\cdot100\text{mL}^{-1}$.

Appendix 08a Acceptable Risk and Tolerable Risk

The concept of Tolerable or Acceptable (pathogen) Risk is touched on in a range of relevant guidelines and documents.

It has in fact no precise definition. This section outlines the key concepts and discussion relevant to the Burwood Beach situation. Overall it appears that what constitutes Tolerable Risk is in a broad
sense a ‘community decision’ which is reached through the kind of process undertaken in the present instance.

Acceptable Risk in the Guidelines

**NHMRC (2008)**
The concept of Tolerable Risk is touched on in the recreation guidelines (NH&MRC, 2008) at the 4 points reproduced below.

**Figure 1.1 Harmonised approach to assessment of risk and management for microbial hazards suitable for any water-related exposure**

1.7 **RISK ASSESSMENT**
These guidelines require that risk be reduced to a tolerable level rather than being eliminated altogether (complete elimination of risk is impossible). For most healthy people, water conforming to the guideline value will pose only a minimal increase in daily risk. However, water conforming to the guidelines may still pose a potential health risk to high-risk user groups such as the very young, the elderly and those with impaired immune systems.
The concept can be seen to be critical as it informs health targets. Acceptability appears to equate in the first instance to general bathing site annual ‘suitability’.

However, what ‘acceptable’ means in practice in the case of short duration high impact events, also identified in the guidelines, is not defined quantitatively probably reflecting the range of hazardous events/exceptional circumstances which each need to be assessed on their own merits.

The main challenge is what constitutes an acceptable likelihood in the case of a hazardous event or ‘exceptional circumstances’. In discussions with NSW Health, the 1% and 5% probability of illness were agreed to be benchmarks for risk assessment discussion purposes regarding hazardous events. What ‘likelihood’ event where these were exceeded though was not resolved.

An option considered was using the product of acceptable illness probability (0.01 or 0.05) and the likelihood of illness under the prevailing enterococci level (0.05 corresponding to the probability of occurrence of the 95th percentile of 40 per 100mL ). But on consideration this path was abandoned because a more detailed review than was possible in the current instance was seen as being needed first.

**WHO (2003)**
The WHO guidelines(World Health Organization, 2003) do not resolve the issue and in fact complicate things further. They explicitly indicate that even the NHMRC(2008) recommendation cannot be taken at face value and risk should be tailored to specific localities (Box 1).

### Table 5.1 Monitoring of microbial alert levels for recreational water

<table>
<thead>
<tr>
<th>Green level Surveillance mode</th>
<th>Amber level Alert mode</th>
<th>Red level Action mode</th>
</tr>
</thead>
<tbody>
<tr>
<td>Monitoring is consistent with the long-term classification, although the water body may be subject to short-term advisories, eg to avoid primary and secondary contact for several days after rain. Continue routine sampling.</td>
<td>Monitoring is not fully consistent with the long-term classification, requiring investigation into the cause of the elevated levels. Increased sampling enables a more accurate assessment of the risks to recreational users. The water body remains subject to short-term advisories, eg to avoid primary and secondary contact for several days after rain.</td>
<td>Monitoring indicates unacceptable risks to recreational users to an extent requiring the local authority and health authorities to warn the public that the water body is considered to be unsuitable for primary and secondary contact.</td>
</tr>
</tbody>
</table>
4.4.5 Adaptation of guideline values to national/local circumstances

There is no universally applicable risk management formula. “Acceptable” or “tolerable” excess disease rates are especially controversial because of the voluntary nature of recreational water exposure and the generally self-limiting nature of the most studied health outcomes (gastroenteritis, respiratory illness). Therefore, assessment of recreational water quality should be interpreted or modified in light of regional and/or local factors. Such factors include the nature and seriousness of local endemic illness, population behaviour, exposure patterns, and sociocultural, economic, environmental and technical aspects, as well as competing health risk from other diseases including those that are not associated with recreational water. From a strictly health perspective, many of the factors that might be taken into account in such an adaptation would often lead to the derivation of stricter standards than those presented in Table 4.7. What signifies an acceptable or tolerable risk is not only a regional or local issue, however, as even within a region or locality children, the elderly and people from lower socioeconomic areas would be expected to be more at risk (Cabella et al., 1979; Priess, 1998).

Box 1. WHO Position on Tolerable Risk (p.72.)

Enhealth (2002)

The Enhealth guidelines (EnHealth Council, 2002) are similarly general in their comments on p. 19 and (in respect to soil contaminants) p. 145 as shown here in Box 2a and 2b.

Risk management

- Information of the community’s concepts of acceptable risk and safety. Community consultation is an integral part of risk management.

A numerical value that would constitute an acceptable level of risk for low-level environmental exposure to carcinogens is not recommended. Whilst there has been considerable debate over the last twenty years about what constitutes an acceptable risk, there is no agreed position internationally on this issue (see Department of the Environment, 1993). The problems of nominating an acceptable level of risk are compounded by the inability of current methods to accurately quantitate risk at low levels of exposure and hence to provide an accurate value that can be compared to ‘an acceptable level of risk’.

Box 2a and 2b. EnHealth (2002) Comments of Acceptable Risk (see p 19 and 145)
Acceptable Risk and QMRA
The reasons for this hesitancy about, and challenge involved in defining Acceptable Risk, is discussed in detail by Hunter and Fewtrell with QMRA specifically in mind (Hunter & Fewtrell, 2001).

Extracts in Box 1 and Box 2 summarise the problem more candidly than the Guidelines though they are quite consistent again with the perception of Acceptable Risk being not easily addressable by ‘commonsense’ or the use of a global measurable water quality guideline statistic. As a result case by case deliberation in light of local circumstances is proposed as the best approach (Box 3).

10.9 IMPLICATIONS FOR INTERNATIONAL GUIDELINES AND NATIONAL REGULATIONS
Although only making up a small input to the harmonised framework, the issue of acceptable risk is an important and extremely complex area. Acceptable risk is very location-specific and for this reason it does not fit within international guidelines, but should play an important role in adapting guidelines to suit national circumstances, where local stakeholder involvement is vital.

Box 1. Overall conclusions

“10 Acceptable Risk
The notion that there is some level of risk that everyone will find acceptable is a difficult idea to reconcile and yet, without such a baseline, how can it ever be possible to set guideline values and standards, given that life can never be risk free? Since zero risk is completely unachievable, this chapter outlines some of the problems of achieving a measure of ‘acceptable’ risk by examining a number of standpoints from which the problem can be approached.

10.1 INTRODUCTION
A number of chapters within this book examine the question of what is risk and how we define it. Risk is generally taken to be the probability of injury, disease, or death under specific circumstances. However, this ‘objective’ measure of risk does not tell the whole story and, in determining acceptability of any particular risk, perceived risk is likely to play a large role.

The following is a list of standpoints that could be used as a basis for determining when a risk is acceptable or, perhaps, tolerable. These will be explored under broad headings. A risk is acceptable when:

• it falls below an arbitrary defined probability
• it falls below some level that is already tolerated
• it falls below an arbitrary defined attributable fraction of total disease burden in the community
• the cost of reducing the risk would exceed the costs saved
• the cost of reducing the risk would exceed the costs saved when the ‘costs of suffering’ are also factored in
• the opportunity costs would be better spent on other, more pressing, public health problems
• public health professionals say it is acceptable
• the general public say it is acceptable (or more likely, do not say it is not)
• politicians say it is acceptable.”

Box 2. Criteria/basis for risk being Acceptable

Given this approach, what are the processes in setting standards for acceptable risk? We would suggest the following systematic approach:
(1) Bring together the group of experts. Ideally this group of experts should represent a broad range of skills and professional backgrounds, and include individuals with skills and expertise in the primary area of interest of the group. In addition, there should also be individuals with broad experience of public health.
(2) Agree the objectives of the group and any constraints to which the group needs to work.
(3) Determine the strength of evidence in support of an association between the environmental factor or indicator under consideration and illness. Make explicit any uncertainties in the data and any assumptions made.
(4) Quantify the impact on the community’s health of the postulated illnesses, again being explicit about assumptions and areas of uncertainty. Consider the issue of particularly susceptible groups.
(5) Model the impact of any proposed change in standards on the community, taking into consideration the wider health, the social and the economic impacts.
(6) Consider whether the resources required to implement changes in any standard are worth the improvement in health.
(cost-utility analysis) and, even if they are, whether the resources required would be more effectively directed at other health goals (opportunity-cost analysis). Again make explicit any assumptions and uncertainties and identify the impact on susceptible groups.

(7) Expose the analytical phase of the standard-setting process to wide scrutiny by stakeholders of every type including pressure groups, expert groups, and industry. In particular seek out views from the wider public health community.

(8) Modify proposals in the light of this consultation exercise.

Box 3. A possible approach for determining Acceptable Risk

**Conclusion**

Encouragingly the approach to dealing with the location specific nature of risk (Box 3) appears to be fully in line with the procedure undertaken in this risk assessment.

Additionally most of the identified bases for defining what risk is acceptable (Box 2) were explored in the case of Burwood Beach.

Our general conclusion thus is that the decisions and decision process reached regarding Burwood Beach Acceptable Risk were in fact obtained by following best practice as defined above.
Appendix 09 Illustrative Example of Risk Characterization via Microbial Risk Probability Calculation

This Appendix illustrates the process of infection and illness risk probability. The example shown in Table 1 is from a risk assessment project undertaken in 2006 on the reuse of highly treated sewage for environmental flows (Khan et al., 2007b; Roser et al., 2006a). The input assumptions are different but the principles used are the same. The process is as follows:

1. A scenario is first developed for the exposure pathway operating and the effectiveness of each stage/barrier is defined quantitatively.
2. Identify the primary wastewater stream (treated wastewater in the example v. treated wastewater in the current assessment);
3. Select a model pathogen (Campylobacter in the example v. a number of pathogens in the current assessment);
4. Estimate its level as a PDF (literature values in the example v. local measurements in the current assessment);
5. Calculate the reduction PDFs for barriers (microfiltration and dilution in the example shown v. environmental inactivation and dilution in the current assessment);
6. Identify the form of exposure and an appropriate consumption rate PDF (drinking in the example v. accidental consumption in the current assessment) to estimate the rate of intake of pathogens;
7. Estimate infection probability and more general health impact.

In Table 2 the inputs and outputs from a Monte Carlo simulated scenario are shown where suboptimal MF-RO removal (MF and RO effectiveness is reduced to a decimal reduction of 2 log_{10} units at a rate equivalent to 5 days per 365 days in a year). In this simulation two models, nominal conditions and malfunction (i.e. Hazardous Event) conditions are run side by side. The final infection statistics are estimated by sampling from one or other model in proportion to the time in that state.

The procedure is shown diagrammatically in Figure 1. This same procedure was used in the current assessment where:

1. The nominal / Baseline conditions corresponded to the dilution+inactivation of effluent or WAS pathogens assuming good mixing sufficient to achieve the typical observed beach water quality.
2. Event conditions corresponded to those 15 minute intervals when modelled microbial particles appeared at the exposure points.
Table 1. Reductions in, and final risk from, *Campylobacter* simulated for Nominal Conditions

<table>
<thead>
<tr>
<th>Stage Barrier</th>
<th>Description</th>
<th>Probability Density Function Type</th>
<th>Units</th>
<th>Output Parameter</th>
<th>Data Source/ Reference</th>
<th>Risk Calculation Output (Average)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Initial Level</td>
<td><em>Campylobacter</em> level in treated effluent modelled using Campylobacter data</td>
<td>Triangular</td>
<td>mpn.L⁻¹</td>
<td>Minimum=30, Mode =1000, Maximum = 10000</td>
<td>(Koenraad et al., 1994)</td>
<td>3676 orgs.L⁻¹</td>
</tr>
<tr>
<td>Microfiltration</td>
<td><em>Campylobacter</em> removal by Microfiltration modelled using Thermotolerant coliforms data</td>
<td>Normal</td>
<td>log₁₀ reduction</td>
<td>Mu=7.916, sigma =2.116 for DR</td>
<td>Singapore Microfiltration thermotolerant coliform PDF data</td>
<td>0.60 orgs.L⁻¹</td>
</tr>
<tr>
<td>Reverse Osmosis</td>
<td><em>Campylobacter</em> removal by Reverse Osmosis modelled using Various indicators data</td>
<td>Point estimate</td>
<td>log₁₀ reduction</td>
<td>Modal value for DR = 6,5</td>
<td>Various</td>
<td>1.90 x10⁻⁷ orgs.L⁻¹</td>
</tr>
<tr>
<td>Dilution</td>
<td>All removal by Dilution at Penrith Weir modelled using particle dilution data from Sydney Water</td>
<td>Point estimate</td>
<td>log₁₀ reduction</td>
<td>Modal value for DR =0.565</td>
<td>Sydney Water Model</td>
<td>5.17 x10⁻⁸ orgs.L⁻¹</td>
</tr>
<tr>
<td>Consumption</td>
<td>Melbourne Overall 0.25 L glass Poisson</td>
<td>Poisson</td>
<td>Litres/day</td>
<td>Gamma = 3.3684</td>
<td>(Mons et al., 2005)</td>
<td>6.71 x10⁻⁸ orgs</td>
</tr>
<tr>
<td>Probability of infection (daily)</td>
<td>Probability of infection based on Outbreak data for <em>Campylobacter jejuni</em></td>
<td>P=1-(1+dose/beta)^α-α</td>
<td>Prob. Of infection</td>
<td>Alpha=0.024, beta = 0.011, Prob. if dose is algorithm output = 6.28E-12, Maximum risk probability, 2.88E-12</td>
<td>(Evans, 1996; Van den Brandhof et al., 2003)</td>
<td>6.71 x10⁻⁸ person⁻¹.day⁻¹</td>
</tr>
<tr>
<td>Probability of infection (annualised)</td>
<td>Annualised Risk Probability</td>
<td>P= 1-(1-Daily Probability)α/365</td>
<td>prob. Of infection</td>
<td>Haas et al. (1999)</td>
<td>2.45x10⁻⁸ person⁻¹.year⁻¹</td>
<td></td>
</tr>
<tr>
<td>Disability Weighted Risk Estimate</td>
<td>Risk expressed as DALY</td>
<td>Point Estimate</td>
<td>DALY per infection</td>
<td>Daly per infection 0.00208</td>
<td>(Haas &amp; Eisenberg, 2001; Havelaar &amp; Melse, 2003; Pruss &amp; Havelaar, 2001 ; Vijgen et al., 2007).</td>
<td>0.051 μDALY</td>
</tr>
</tbody>
</table>

1. Simulation corresponds to dry weather (80th percentile) flow.
<table>
<thead>
<tr>
<th>Stage Barrier</th>
<th>Description</th>
<th>Probability Density Function Type</th>
<th>Units</th>
<th>Output Parameter</th>
<th>Data Source/ Reference</th>
<th>% of Simulations</th>
<th>Risk Calculation Output (Average)</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Initial Level</td>
<td><em>Campylobacter</em> level in treated effluent modelled using Campylobacter data</td>
<td>Triangular distribution</td>
<td>mpu.L⁻¹</td>
<td>Minimum=30, Mode =1000, Maximum = 10000</td>
<td>(Koenraad <em>et al</em>., 1994)</td>
<td>100</td>
<td>3676 orgs.L⁻¹</td>
<td></td>
</tr>
<tr>
<td>Microfiltration</td>
<td><em>Campylobacter</em> nominal removal by Microfiltration modelled using Thermotolerant coliforms data</td>
<td>Normal distribution</td>
<td>log₁₀ reduction</td>
<td>Mu=7.916, sigma =2.116 for DR THC removal PDF</td>
<td>Singapore MF THC removal PDF</td>
<td>98.5</td>
<td>0.60 orgs.L⁻¹</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Campylobacter</em> suboptimal removal by Microfiltration modelled using Thermotolerant coliforms data</td>
<td>Point Estimate</td>
<td>log₁₀ reduction</td>
<td>Modal value for DR = 2</td>
<td>(Kitis <em>et al</em>., 2003)</td>
<td>1.5</td>
<td>36.8 orgs.L⁻¹</td>
<td></td>
</tr>
<tr>
<td>Reverse Osmosis</td>
<td><em>Campylobacter</em> nominal removal by Reverse Osmosis modelled using Various indicators data</td>
<td>Point estimate</td>
<td>log₁₀ reduction</td>
<td>Modal value for DR = 6.5</td>
<td>Various</td>
<td>98.5</td>
<td>1.90 x10⁻¹ orgs.L⁻¹</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Campylobacter</em> suboptimal removal by Reverse Osmosis modelled using Various indicators data</td>
<td>Point estimate</td>
<td>log₁₀ reduction</td>
<td>Modal value for DR = 2</td>
<td>Kitis <em>et al</em>., (2003)</td>
<td>1.5</td>
<td>3.68 x10⁻¹ orgs.L⁻¹</td>
<td></td>
</tr>
<tr>
<td>Dilution</td>
<td>All removal by dilution at Penrith Weir modelled using particle dilution data from Sydney Water</td>
<td>Point estimate</td>
<td>log₁₀ reduction</td>
<td>Modal value for DR =0.565</td>
<td>Sydney Water Model</td>
<td>98.5</td>
<td>5.17 x10⁻⁶ orgs.L⁻¹ nominal conditions</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Consumption Melbourne Overall 0.25 L glass Poisson</td>
<td>Poisson distribution</td>
<td>Litres/day</td>
<td>Gamma = 3.3684</td>
<td>(Mons <em>et al</em>., 2005)</td>
<td>98.5</td>
<td>6.71 x10⁻⁸ orgs nominal conditions</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Probability of infection (daily)</td>
<td>P=1-(1+dose/beta)^alpha</td>
<td>Prob. of infection</td>
<td>Alpha=0.024, beta = 0.011, Prob. if dose is algorithm output = 6.28E-12, Maximum risk probability, 2.88E-12</td>
<td>Van den Brandhof <em>et al</em>., (2003) Evans <em>et al</em>., (1996)</td>
<td>100</td>
<td>3.03 x10⁻², person⁻¹.day⁻¹ combined simulation</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Probability of infection (annualised)</td>
<td>P= 1-(1-Daily Probability)³65</td>
<td>Prob. of infection</td>
<td>Haas <em>et al</em>., (1999)</td>
<td></td>
<td>100</td>
<td>1.05x10⁻⁵ person⁻¹.year⁻¹ combined simulation</td>
<td></td>
</tr>
<tr>
<td>Disability</td>
<td>Weighted Risk Estimate</td>
<td>Point Estimate</td>
<td>DALY per infection</td>
<td>DALY per infection 0.00208</td>
<td>(Haas &amp; Eisenberg, 2001; Havelaar &amp; Melse, 2003; Pruss &amp; Havelaar, 2001 ; Vijgen <em>et al</em>., 2007).</td>
<td>100</td>
<td>217 μDALYs combined probability</td>
<td></td>
</tr>
</tbody>
</table>
Figure 1. Procedure for Simulating the Risk Arising from Baseline+ **Hazardous Event** Conditions
Appendix 10 Seasonality, Outbreaks and Hazardous Events

This risk assessment has been focused on contaminant ‘plume’ Hazardous Events. Two other types of ‘Hazardous Events’ are also of concern to NSW Health and the draft report’s independent reviewer, which are associated with elevate pathogen levels, and hence risk:

1. ‘Seasonality’;
2. ‘Outbreaks’.

‘Seasonality’ in water borne disease outbreaks has often been identified as a factor to consider in risk assessment (Ashbolt et al., 2001). This wording suggests the classic four climatic seasons of spring, summer, autumn and winter when for reasons partly understood there are peaks in disease rates, as for example the ‘winter’ (swine) influenza peak discussed at the time of writing (note that human influenza is not currently considered a sewage borne pathogen).

The desirability of considering ‘Outbreaks’ is similarly self-evident because such periods are by definition periods on increased risk when there may be abnormally high source levels of pathogens, breakdowns in the barriers separating pathogen sources from exposed populations, enhanced intake, or increased exposure to a vulnerable sub-populations. The term ‘Outbreaks is widely used to refer to sudden unexpected outbreaks of disease in the community like the ‘Walkerton Outbreak’.

Incorporating ‘Seasonal’ and ‘Outbreak’ associated pathogen level peaks into the current risk assessment in a precise way has not been undertaken formally, however. This is because the complexity of these phenomena is so great compared to the limited amount of definitive data available that an analysis comparable to that undertaken for the effluent and WAS plumes, did not appear practical at the present time. In the meantime:

1. Note that seasonality has been addressed in this QMRA as follows:
   a. Behaviour of plumes in summer compared to winter has been modelled;
   b. The greatest disease risk overall is the ‘Seasonal’ risk arising from the general bathing population using the beaches in the summer months.
   c. This latter population probably includes children and the elderly more than the surfing population who would be expected to comprise mainly young to middle age adults who are relatively healthy.
2. In the main assessment we have included selected ‘Sensitivity analyses’ addressing the question of what would be the risks ‘what if’ selected pathogens were higher in level by one, two or three orders of magnitude for certain more definable scenarios. The risk estimates made can be applied to seasonal and outbreak risk;
3. Because sensitivity analyses of outbreak / seasonal peaks of pathogens are necessarily conceptual/speculative they have been discussed further in the final Uncertainties section to highlight the provisional nature of the current assessment;
4. This Appendix outlines the information identified on seasonality and disease outbreaks due to water borne pathogens in natural bathing waters for future consideration on how data constraints might be addressed.

Why Seasonality and Outbreaks cannot be Accounted for fully at this Time

From out review of the literature useful/reliable Scenario modelling of the risks posed by ‘Seasonality’ and ‘Outbreaks’ based on the current literature data appeared logistically impracticable (apart from general scoping and inclusion of ‘Sensitivity’ scenarios noted in the previous section) for the following reasons:

1. The data available (Table 1) is limited. There is substantial statistical data on bathing outbreaks (Craun et al., 2005; Sinclair et al., 2009; Westrell, 2004), but not on the
underlying drivers and source impact/pathogen levels. By the time outbreaks are identified and analysed the environment is often altered. So while some information is currently available for a range of speculative/conceptual analyses we did not consider it sufficient for reliable QMRA.

2. ‘Outbreaks’ and ‘Seasonality’ are not distinct but rather overlap in the literature and appear to be confused semantically and conceptually. An example is the concept of ‘Season’. This can refer to the four strikingly different annual climate periods characteristic of the northern cool temperate climatic zones. Or it can refer to wet and dry period whose overlap with temperature driven seasons varies according to region. The extent of climatic change characteristic of different seasons varies. It is striking in northern temperate climate zones but it is less pronounced in warmer climates more comparable to New South Wales. In the tropics such seasonality is even less marked especially compared to the wet/dry climate cycles which itself are regionalised. Temperature is not a simple indicator due to the confounding influence of ocean/continental weather patterns and elevation (The equatorial Andes have a year round climate more similar to northern New Zealand than the adjacent Amazon). Europe and North America are not climatically uniform either with large parts having a more temperate ‘Mediterranean climate than the colder/alpine areas where much work on the effect seasons has been undertaken (Sweden, Michigan).

3. Available data on endemic pathogen levels in water and wastewater with ‘Season’ is limited, generally confined to 6 or 12 month periods.

4. Seasonal disease peaks are often an outcome of changes in community behaviour e.g. summer recreation activities associated with water contact and this issue was largely covered in the current assessment.

5. The term ‘Seasonality’ is used for a variety of different phenomena e.g. seasonality in disease causing organisms disease, occurrence of pathogens in stools.

Studies of Seasonality
The data available at best indicate the factors underlying ‘seasonal variability’ are multiple. Risk varies not only with the diversity and abundance of pathogens but also with seasonally correlated factors such as temperature, UV, precipitation, occupancy of area etc. (Griffin et al., 1999). It probably also depends on the health status of the populations under consideration and to a lesser extent local agriculture.

In the past a number of authors have noted that water borne disease ‘seasonality’ has not been satisfactorily characterized (Gibson et al., 1998). To judge by the information available now this is still the case. In respect to recreation specifically it has been noted that “Because most of the outbreaks occurred in countries above the 40° latitude, most investigations in this review can be indicated as summer outbreaks (i.e. when people recreate). This illustrates how seasonality reflects behaviour (i.e. preference for warm water by large numbers of people) (Sinclair et al., 2009) as much as truly seasonal peaks in pathogen occurrence. Other complications making it difficult to construct rules about ‘seasonality’ suited to QMRA Scenario construction include:

1. The picture is not consistent. Some authors have found seasonality, some have not;
2. Pathogen mobilization into waterways is driven by run-off depending on local rainfall pattern may be seasonal;
3. The role of sunlight and the effects of local latitude and climate are understood conceptually but not yet comprehensively in a quantitative fashion (our studies on the effect of sunlight on plume pathogens are only one aspect but the illustrate the difficult and complex task of accounting to this factor alone);
4. Viruses appear to be the critical pathogen group. However virology assays are notoriously variable in quality over time and between laboratories as judged by inter-laboratory trials e.g. (Sellwood & McDermott, 1993; Sellwood & Shore, 1997).
5. Interpretation of virus data is confounded by uncertainty about:
   a. How to relate real time PCR copy number estimates to infectious units and infection probability;
   b. The extent to which cultural methods can detect different virus strains within a particular group (dealt with further in the Uncertainty section of the main report text);
   c. Viral assays are moving from culture based studies to PCR based assays. Only rarely are the two done in parallel and this has shown that PCR which can yield very different estimates of virus numbers to culture methods (He & Jiang, 2005). It is unclear how well these contrasting data sets follow one another but it probably depends on the pathogen of interest and the sample type.

6. Epidemiology studies only identify elevated disease above a certain threshold (e.g. Westrell, 2004).

7. Water borne pathogen outbreaks may have different origins e.g. an infected individual or contaminated food who may be hard to identify in follow-up work.

From our examination of the literature (Table 1) it appears that there are some good location/pathogen specific data on the occurrence of seasonal peaks and outbreaks. However, it is very diverse and there is still insufficient information in our opinion to establish QMRA scenarios for the Burwood Beach system in the same manner as has been done with plume events beyond the Uncertainty analysis undertaken.

Table 1. Reports on the seasonality of water borne pathogens or outbreaks likely caused by them

<table>
<thead>
<tr>
<th>Matrix/Sample</th>
<th>Pathogen/Indicator</th>
<th>Seasonality</th>
<th>Assay</th>
<th>Comments</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Oysters</td>
<td>Adenovirus</td>
<td>Not seen</td>
<td>PCR</td>
<td></td>
<td>(Pina et al., 1998)</td>
</tr>
<tr>
<td></td>
<td>Hepatitis A Virus</td>
<td>Not seen</td>
<td>PCR</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Enteroviruses</td>
<td>Cooler months</td>
<td>PCR</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wastewater</td>
<td>Norovirus GGI</td>
<td>Highest during summer</td>
<td>PCR</td>
<td></td>
<td>(Nordgren et al., 2009)</td>
</tr>
<tr>
<td></td>
<td>Norovirus GGII</td>
<td>Highest during winter</td>
<td>PCR</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Diarrheal Samples</td>
<td>Entero-pathogens</td>
<td>Rainy season</td>
<td>Culture</td>
<td>Increase factor of about 2</td>
<td>(Ono et al., 2001)</td>
</tr>
<tr>
<td>generally</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wastewater</td>
<td>Campylobacter</td>
<td>Later spring or early summer or not at all</td>
<td>Mainly culture</td>
<td>Review – unclear how reliable identification is : see (Diergaardt et al., 2003 ; Diergaardt et al., 2004)</td>
<td>(Skelly &amp; Weinstein, 2003)</td>
</tr>
<tr>
<td>Sludge</td>
<td>Campylobacter</td>
<td>April to June northern hemisphere</td>
<td>Culture</td>
<td>Very distinct peaks</td>
<td>(Jones et al., 1990)</td>
</tr>
<tr>
<td>River sediments</td>
<td>(Indicators)</td>
<td>Tourist season</td>
<td>Culture</td>
<td></td>
<td>(Donovan et al., 2008)</td>
</tr>
<tr>
<td>Wastewater</td>
<td>Giardia</td>
<td>Unclear</td>
<td>Microscopy</td>
<td>4 reports cited of which 2 said yes and 2 no</td>
<td>(Caccio et al., 2003)</td>
</tr>
<tr>
<td>Biosolids</td>
<td>Reovirus</td>
<td>Unclear</td>
<td>Various</td>
<td>Review paper</td>
<td>(Sidhu &amp; Toze, 2009)</td>
</tr>
<tr>
<td></td>
<td>Adenovirus</td>
<td>Unclear</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Polyomavirus</td>
<td>Unclear</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Somatic coliphage</td>
<td>Unclear</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Bacterial indicators</td>
<td>No</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Giardia</td>
<td>No</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Oyster gastroenteritis</td>
<td>Vibrio vulnificus</td>
<td>Midsummer</td>
<td>Culture</td>
<td>Consumption peak</td>
<td>(Graczyk &amp; Schwab, 2000)</td>
</tr>
<tr>
<td>Drinking source waters</td>
<td>Torque Teno Virus</td>
<td>No</td>
<td>PCR and Cell culture</td>
<td>Proposed indicator</td>
<td>(Griffin, 2009)</td>
</tr>
<tr>
<td>Matrix/Sample</td>
<td>Pathogen/Indicator</td>
<td>Seasonality</td>
<td>Assay</td>
<td>Comments</td>
<td>Reference</td>
</tr>
<tr>
<td>--------------</td>
<td>-------------------</td>
<td>-------------</td>
<td>-------</td>
<td>----------</td>
<td>-----------</td>
</tr>
<tr>
<td>Environmental water</td>
<td>Calciviruses</td>
<td>Winter</td>
<td>PCR</td>
<td>Made up of Norvirus, Sapporovirus and two others. More research needed.</td>
<td>(Schaub &amp; Oshiro, 2000)</td>
</tr>
<tr>
<td>Wetland discharge</td>
<td>Coliforms</td>
<td>Winter</td>
<td>culture</td>
<td>Lower performance</td>
<td>(Thurston et al., 2001)</td>
</tr>
<tr>
<td>Wastewater</td>
<td>Enterovirus</td>
<td>No/Winter</td>
<td>Culture</td>
<td>Depends on treatment plant</td>
<td>(Petrinca et al., 2009)</td>
</tr>
<tr>
<td>Disease incidence and organism occurrence</td>
<td>Cryptosporidium</td>
<td>Rainfall</td>
<td>clinical</td>
<td>Cited in</td>
<td>(Rose et al., 2001)</td>
</tr>
<tr>
<td></td>
<td>Acanthamoeba keratitis</td>
<td>June and Nov.</td>
<td>Clinical and surveys</td>
<td>Review</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Diarrheal agents</td>
<td>El Nino</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>All agents</td>
<td>Extreme precipitation</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Cyclosporiasis</td>
<td>Summer</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wastewater</td>
<td>Adenovirus</td>
<td>None</td>
<td>PCR</td>
<td></td>
<td>(Carducci et al., 2008)</td>
</tr>
<tr>
<td>Urban rivers</td>
<td>Indicators and coliphages</td>
<td>Winter/rainfall</td>
<td>Culture</td>
<td></td>
<td>(Choi &amp; Jiang, 2005)</td>
</tr>
<tr>
<td></td>
<td>Adenoviruses and enteroviruses</td>
<td>None</td>
<td>PCR</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Coastal Waters</td>
<td>Hepatitis E Virus</td>
<td>Winter</td>
<td>PCR</td>
<td>Range of complicating environmental variables</td>
<td>(Fong et al., 2005)</td>
</tr>
<tr>
<td></td>
<td>Human Adenovirus</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wastewater</td>
<td>Rotavirus</td>
<td>None</td>
<td>Culture</td>
<td>Contrasts with seasonality elsewhere</td>
<td>(Bosch et al., 1988b)</td>
</tr>
<tr>
<td>River water</td>
<td>Norovirus</td>
<td>Winter</td>
<td>PCR</td>
<td>Factor of 6 higher.</td>
<td>(Westrell et al., 2006)</td>
</tr>
<tr>
<td>Various</td>
<td>Adenovirus</td>
<td>spring and summer</td>
<td></td>
<td></td>
<td>White et al. 1996 cited by (Sinclair et al., 2009)</td>
</tr>
<tr>
<td></td>
<td>Enterovirus</td>
<td></td>
<td></td>
<td>Survey</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Cryptosporidiosis</td>
<td>High rainfall</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sewage</td>
<td>Adenovirus</td>
<td>Little variability</td>
<td></td>
<td></td>
<td>(McOliver et al., 2009).</td>
</tr>
<tr>
<td>Sewage before and after treatment</td>
<td>E. coli and thermotolerant coliforms</td>
<td>Peak in early autumn</td>
<td>Culture and microscopy</td>
<td>Between sample variation unusually small</td>
<td>(Payment et al., 2001)</td>
</tr>
<tr>
<td></td>
<td>C. perfringens and faecal streptococci</td>
<td>No variation of over spring - autumn</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Giardia</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Cryptosporidium enteroviruses</td>
<td>Clear peak early autumn</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Notes:
1. Seasonality reported is a mixture primary data, review information, author interpretations.

**Outbreaks**
The literature seldom doesn’t distinguish between Seasonality and Outbreaks well. Pathogen monitoring data is generally insufficient to indicate clear seasonal trends. Data linking pathogen level to community disease levels appears lacking except where outbreaks are followed up. Even in this case where Norovirus outbreaks were followed up in a timely fashion, detection of pathogens in natural waters proved impossible (e.g. Jones et al., 2009).

Overlap also occurs between the concepts of Seasonality and Outbreaks. This is illustrated in Figure 2 of Rose et al. (Rose et al., 2001) where the statistics which can be viewed as indicating outbreaks
and/or seasonality depending on definition. Put another way there can be seasonal spikes in disease which if great enough can be interpreted as outbreaks, strict definitions aside. Similarly ‘Outbreaks’ may be triggered by seasonal factors such as heavy rainfall leading to overloaded sewers and contaminated rivers.

This said distinct outbreaks occur periodically which may be associated with the seasons while not being strictly seasonal as such. Sinclair et al. have very recently published a survey on this topic of bathing water outbreaks which necessarily tend to be in the summer. Their review indicates that Norovirus appears responsible for 45% of outbreak related recreational disease (Sinclair et al., 2009). This suggests that ideally Norovirus would have been the preferred replacement model virus for Rotavirus. However:

1. It was unclear which Norovirus subtype should be screened for;
2. It is unclear how PCR copy numbers relate to infectious doses or viable particle numbers;
3. Although Norovirus appears to be developing into the virus index of choice the only local laboratory available was already committed to an extensive work program (Peter White pers. com.).

After Norovirus, Adenovirus appeared to be the most important cause of recreation derived disease (Sinclair et al., 2009) and its incidence is only marginally lower. This indicated that the selection of Adenovirus to Rotavirus was the best option available.
Appendix 11 Water Quality and Hydrological Strategic Monitoring and Project Implementation

Introduction
A survey of wastewater and WAS quality was undertaken by local contract water quality laboratories as well as HWC aimed at providing sufficient data to estimate probability density function (PDF) coefficients defining the typical Baseline quality of the raw screened sewage, secondary treated effluent which could be used subsequently in QMRA modelling.

The index pathogens analysed for were:
1. Cryptosporidium spp.,
2. Campylobacter spp.,
3. Rotavirus.
4. Giardia lamblia
5. Adenovirus
6. (Enterococci).

To assess the extent to which pathogen levels measured at ca 9 am were likely to be representative of levels over the full course of the day intensive indicator monitoring was also undertaken to assess variation diurnally and over the course of the survey.

To complement this work data on concurrent waste stream flows was also collected to allow estimation of pathogen loads (organisms discharged per unit time) and determine:
1. Whether the measurements collected reflected those of normal dry weather flows or were impacted by wet weather surges;
2. The reduction in pathogen numbers achieved by the secondary treatment processes;
3. The extent to which pathogens were concentrated in the WAS and partitioned between the two streams;
4. Whether the pathogen level PDFs were sufficiently representative to be used direct in the QMRA or flow weighted average means would be more precise;
5. Whether it was preferable to include simulation of WWTP treatment in the QMRA models or simply use the discharge level estimates.

Whether the WAS posed special pathogen related risk was addressed by:
1. Separate analysis of its pathogen content;
2. Comparison of this content data with that in the other stream for consistency which would suggest the levels were being underestimated;
3. Determination of likely pathogen fate and transport by hydraulic modelling and QMRA of WAS as well as the secondary effluent streams;
4. Modelling of 2030 as well as the 2007 hydraulic loads to estimate the increase in risk due to increased discharge rates;
5. Experimental measurements of WAS v. secondary effluent indicator inactivation.

The issue of variable pathogen survival was addressed by:
1. Modelling inactivation rates covering rates ranging from the most rapid reasonably conceivable to the most conservative (dilution only);
2. The experiments on indicator and inactivation to determine at what rates this was likely to occur and how the rates varied between WAS and secondary effluent;
3. Measurement of ocean water transmissivity, which would likely influence solar radiation driven inactivation.
As with the quantity of data collected the reviewer was concerned with the “low level of model calibration”. We have addressed this by providing further detail on the hydraulic model calibration, use and origins. Further to this the following should be noted:

- The aim of the assessment was not primarily to provide large amounts of data for a greenfields project but to inform decision making as to whether the WWTP upgrade was sufficient or more works (or potentially further assessment) were needed.
- The general approach and scale was endorsed by regulators at the assessment commencement.
- Risk assessment must be iterative and this has been allowed for by the review process and identifying knowledge gaps. The option to address the latter further is left to the decision makers as is appropriate.
- The limitations of the study have been extensively documented.
- Though the reviewer suggested that world wide practice was more detailed we are unaware of full QMRA being applied to coastal bathing waters to the extent that we have applied it. Put another way the current study like all studies has both strengths and limitations and cannot cover all issues, though it can facilitate further review by transparency and has endeavoured to do this.
- The existing data suggested that beach water was of a high quality and a total reassessment of risks was hard to justify in the first instance as a results.

Methods

System Description

Exposure pathways to be considered for exposure pathway assessment were developed and are shown in Figure 1 as these are central to risk estimation and exposure scenario construction. These are in line with the original pathways considered in the WRL report (Glamore et al., 2008).
Water Quality Analysis Survey

Survey Outline

It was initially planned that the levels of three reference or index pathogens would be measured. The three selected were:

- Cryptosporidium parvum/hominis
- Campylobacter jejuni
- Rotavirus

Giardia spp. (assume G. lamblia) were also being measured because their enumeration is straightforward to undertake when Cryptosporidium is measured and based on another concurrent study (Roser & Ashbolt, 2008) it appears that they are likely to be present in much higher numbers than Cryptosporidium spp. and pose an even higher risk.

Early in the monitoring program it appeared that there was little evidence of Rotavirus in the assays used by Environmental pathogens. A further contingency was the potential to measure Adenovirus level in the event there are few rotavirus.

Enterococci were also included as a pathogen ‘surrogate’ in this group of measurements when it was realized that such data could be used in an empirical dose response relationship to estimate total gastrointestinal illness probability.
Along with pathogen monitoring indicators were measured in the waste-streams over the course of three 24 hour periods so as to:

4. assess treatment process effectiveness for different microbial groups;
   • assess how variable levels of micro-organisms are;
   • provide a separate supporting data on how microorganisms behave generally.

A provisional timeline proposed at the commencement for the overall program was as follows:

• 1 month for preparation;
• 3 months for data collection;
• 1 month risk modelling with final data;
• 1 month for report preparation;
• 1 month contingency to allow for Xmas disruptions occurring in the latter part of the program.

Logistics Issues

Some delays were experienced due to:

1. The additional time taken to compile the water quality laboratory data;
2. Unexpected delays in assessing F-RNA bacteriophage die-off in the microcosms;
3. Completing the full suite of hydraulic modelling simulations;
4. Completing the integrated modelling;
5. A request by HWC for an interim report on risk probability estimates and some minor additions to the data analysis program (analysis of historical enterococci data).

Model run times were maximized within the limits of the hardwater. The reason for this long hydraulic modelling time was as follows:

• The integrated hydrodynamic model involved simulating in each scenario the fate (transport and inactivation) of each of a very large number of particles (1.9 billion in the final scenarios) over an extended period (2 days in the case of conservative particle, 7 days in the case of those inactivated) in three dimensions along the coastline divided into a grid with cells of only 7500 m³, and recording changes in (pathogen) particle numbers as they were diluted, and their mass, as they are inactivated.

• Critical to the success of the modelling was the ability to track relatively highly diluted particles because even highly diluted effluent and WAS posed a potential risk because of the high load of pathogens. Programmatically this was straightforward. But to gain a relatively small ten fold increase in sensitivity over previous modelling was difficult. A modest 10 fold increase in the number of particles tracked in the model would require a corresponding 10 fold increase in scenario running time compared to previous years. Fortunately workstation computing speeds have increased greatly in the past few years so tracking an increased number of particles compared to previous modelling appeared feasible despite the large number of scenarios to be modelled (n=256). More than this though was unworkable.

• A degree of compromise was still necessary and in the end the number of particles simulated in the model was sufficient to detect/follow secondary effluent and WAS diluted in the bathing zones by a factor of up to ca $10^4$.

Another challenge was to find an efficient method for communicating infection/illness probability. The integration of QMRA and hydraulic modelling was anticipated to generate a range of risk estimates any of which were of potential interest to the project stakeholders. The literature did not offer any satisfactory model. Rather the tendency was to choose one or two arbitrary percentiles to report against. This was seen as satisfactory for these journal papers but not in the present instance.
Water Quality Analyses

Contract Analyses
Wastewater and WAS samples were analysed by:
1. Hunter Water Corporation Laboratory (Protozoa and bacterial indicators);
2. Australian Water Quality Centre (Campylobacter);
3. EML (F-RNA bacteriophage);
4. Environmental Pathogens (Rotavirus and Adenovirus).

Quality control samples (replicates, spiked microorganisms, collection blanks) were included where practicable.

Only ‘Dry Weather’ samples were collected and the major rainfall surcharge events were avoided in sampling.

Weekly Pathogen Monitoring
Due to the time constraints the monitoring would take place over a period of 3 months. The index pathogens plus enterococci monitoring had the following attributes:
- 12 samples were collected for each of the three waste streams at the rate of ca one per week except where there had been heavy rain in the two days before collection;
- In addition there were 4 to 6 QA/QC samples taken per analyte;
- The total number of samples analysed was ca 36 + 18 = 54 per analyte.

Diurnal Variation of Indicator Numbers
For the indicators (E. coli, enterococci, C. perfringens, F-RNA coliphage) diurnal sampling was undertaken using an autosampler at intervals of ca 1 month:
- For each wastestream, 6 samples were collected over the course of three 24 hour periods (n= 18);
- These were complemented by QA/QC samples (3 types), analysed for indicators (total analyses per analyte = 30 measurements.

Carryover was assessed by triggering the autosampler to collect a sample from a 10L container of sterilized deinonized water into a bottle in the autosampler located in line next to the actual samples. This was similar to work done previously on sewage samples in a study of river water where a carryover of 2% was observed (Roser & Ashbolt, 2007).

Work Implementation
Timing of the work is shown in Table 1.

Table 1. Summary of WWTP Sampling Undertaken

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<th>Analytes</th>
<th>Experiment Group</th>
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**WWTP Hydraulics**
Hunter Water Corporation provided 30 minute and daily total flows for screened primary sewage, secondary treated effluent and WAS as well as daily rainfall data for the period of the water quality monitoring survey.

The daily rainfall data measurements were taken as reflecting general Newcastle rainfall as well as rainfall in the Burwood Beach area.

This data was used for the following:
1. Assessment of the diurnal flow patterns to determine where in the hydrograph the sample collections tended to occur;
2. Calculations of pathogen and indicator loads into and out of the WWTP, partitioning into WAS and the total removal of pathogens achieved by the WWTP processes;
3. Identification of when rainfall event surges occurred so as to:
   a. Identify which flows could be considered ‘dry weather’
   b. Estimate dry weather flows following removal of high flow period data.

Ancillary Analyses

The following was undertaken to support the risk modelling:
1. As part of the verification work measurements of seawater transmissivity to solar radiation were undertaken.
2. Concurrently seawater, treated water and WAS were collected and analysed at the laboratory to determine the transmissivity of different dilutions of wastewater and WAS.
3. The literature was surveyed for data on the fate/reduction of pathogens in treated effluent and WAS.
4. Inactivation rates were measured to determine if the coastal seawater inactivation modelling assumptions ranging from high solar radiation to conservative covered reasonably the full range of possible inactivation.

WAS is an unusual material upon which to undertake risk assessment. So testing of WAS employed methods used by the contracted laboratories for assaying sludge and biosolids. How reliable these methods were was not completely clear.

Direct estimation of inactivation of pathogens within diluted WAS in the field was impractical within the three month main experimental time frame. So the following approach was taken to better understand/quantify the rate at which pathogens in WAS are inactivated:
1. The rates at which indicators are inactivated in a model microcosm system (Davies et al., 2008b) were estimated;
2. Pathogen inactivation rates for aquatic environments were identified in the literature and used to interpret modelling results in the case of both secondary effluent and WAS.

Data Management and Analysis

Database and it format.

All data were assembled in an MS Access database to facilitate calculations, data filtering, summarisation, editing and data management. Prior to entry all data were screened for quality and if necessary edited with MS Excel (especially in the case of water quality data).

To avoid confusion the arithmetic symbols were replaced with letter codes:
- EQ : ‘=’
- CA : circa (treated as equal to) for calculation purposes
- LT : Less than;
- LE : Less than or equal to;
• GT : Greater than;
• GE : Greater than or equal to.

The main tables generated appeared as follows:

- **Hydro daily** (daily flow and rainfall data for the WWTP) e.g.

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- **Hydro30min** (30 minute flow data) e.g.;

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- **Quality** (each microbial water quality) e.g.

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<th>Time</th>
<th>Material</th>
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</tbody>
</table>

- **WRL Model Runs** (Hydraulic Model Run Outputs e.g.)

<table>
<thead>
<tr>
<th>ID</th>
<th>SEASON</th>
<th>DISCHARGE</th>
<th>GE_TYPE</th>
<th>YEAR</th>
<th>DOCKTO_N</th>
<th>Date_Time</th>
<th>O_ PARTICLE</th>
<th>MEDIAN_ MASS</th>
<th>90TH%ILE_ Mass</th>
<th>99TH%ILE_ Mass</th>
<th>SUM_ OF_ MASS</th>
<th>Median of Travel Time</th>
</tr>
</thead>
<tbody>
<tr>
<td>156013</td>
<td>Summer</td>
<td>WAS 3</td>
<td>2007</td>
<td>1</td>
<td>9/02/2007 8:45:00 AM</td>
<td>2.609E+07</td>
<td>6.09E+07</td>
<td>6.09E+07</td>
<td>1.23E+08</td>
<td>47400</td>
<td></td>
<td></td>
</tr>
<tr>
<td>167799</td>
<td>Summer</td>
<td>WAS 3</td>
<td>2007</td>
<td>2</td>
<td>15/03/2007 10:15:00 AM</td>
<td>7280000</td>
<td>7280000</td>
<td>7280000</td>
<td>0</td>
<td>59400</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Further explanation of these outputs is provided below.

- **SimRuns 2_1** (statistics generated by each QMRA model scenario for QC, reporting and in the event of auditing).

(There are too many fields to show here but an example of a single record is shown in Appendix 17 Example of Part of Simruns 2_1 Record Table. Note that the information stored includes all critical model input details and percentile outputs).
**Microbial data adjustment**

Where possible and appropriate pathogen level data were ‘adjusted’ based on the recovery of spikes. This was done in the case of:

1. Cryptosporidium;
2. Giardia;
3. Adenovirus;
4. Rotavirus;
5. F-RNA bacteriophage.

Reports with microbial levels as a result have term ‘adjusted’ where this has been done.

Cryptosporidium and Giardia counts are only reported as total counts as the laboratory judged the contents to difficult to assess as being ‘confirmed’. This was not seen as a major problem because these protozoa were necessarily freshly excreted and the counts of them were necessarily conservative.

Recovery of enterococci and E. coli were judged to be sufficiently high that adjustment was not needed to the counts.

The analytical laboratory reported that no reliable estimate of recovery was possible with their assay method. Estimation of C. perfringens recovery was apparently not possible either.

Where the estimated pathogen or indicator level was less than the detection limit, the half detection limit value was calculated. The log_{10} of each value was also calculated in all cases as it was assumed that as has been found in the past for similar data it fitted approximately to a log normal PDF.

In the final data sets the use of the half detection limit method was limited and it was not used exclusively. In the case of the new pathogen and indicator data the PDFs actually used employed the Palisade coefficient estimation tool (next section) and the functions calculated using half detection limit substitution values were used only as a check that the other PDFs were of the correct order of magnitude.

In the case of the historical data percentiles were estimated not only using the half detection limit substitutions (approximately 50% of each enterococci data set) but directly from the data sets which each comprised over 200 measurements. As a result actual population PDFs were tabulated. The 95th percentiles estimated using different methods were found to be within a factor of 2 of one another and this different did not affect the overall picture of risk obtained from this data.

**Estimation of Probability Density Function Coefficients**

In environmental sampling of pathogens it had previously been found that the data were generally consistent with a log_{10} normal PDF (Roser & Ashbolt, 2007). So for QMRA purposes the initial (source) pathogen levels were assumed to follow this distribution form. Checks of complete data sets for goodness of fit using the Kolmogorov-Smirnov Test (Massey, 1951) indicated this was the case.

Where data sets were complete the mean (of the logarithms) and standard deviation (SD) were calculated using normal MS Excel functions. Where data was censored (i.e. many values below the detection limit):

1. the data was assumed to follow a log normal distribution and sorted in descending order;
2. percentiles for above detection limit values were calculated using the Excel Percent Rank function;
3. the resulting value/percentile pairs were then analysed using the @Risk 4.5 PDF coefficient estimation tool (Palisade) and obtain estimates of the (log) normal mean and standard deviation.

This general extrapolation method is known as the robust parametric method (El-Shaarawi, 1989) and we have found it previously to provide a satisfactory estimate of log normal distribution (Roser & Ashbolt, 2007).

In the end data extrapolation was largely unnecessary and would not have impacted on the risk estimates. The main analytes whose data was used were enterococci, Giardia and Adenovirus. The first two included no below detection measurements. In the case of Adenovirus there were only two measurements which were below detection limit out of a total of 36 waste stream assays.

Results and Discussion

Microbial Levels Sewage

An extensive set of both the primary pathogen and supporting indicator data were obtained in the course of the survey (Appendix 23 Wastewater and WAS Quality). QA/QC assessments indicated that in most cases the data was reliable or alternatively where problems occurred (Table 2). Most problematically the validity of the rotavirus assay tests was unclear as few positives were detected. As the virus samples were still available and it was possible to assess the levels of this index pathogen its PDFs were estimated instead.

Table 2. Overall Success in Microbial Data Acquisition

<table>
<thead>
<tr>
<th>Sampling Activity</th>
<th>Analyte</th>
<th>Suitability of Data for Estimating PDF Coefficients</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Weekly Sampling</td>
<td>Cryptosporidium total adjusted</td>
<td>Sufficient</td>
<td>Cas 50% of counts below detection but data still sufficient to estimate PDF coefficients. Recovery was not ideal but estimated levels were comparable to previous studies.</td>
</tr>
<tr>
<td></td>
<td>Giardia total adjusted</td>
<td>Sufficient</td>
<td>No non-detects. Sufficient data to estimate PDF coefficients. Spike recovery not ideal.</td>
</tr>
<tr>
<td></td>
<td>enterococci</td>
<td>Good</td>
<td>High frequency of good count data, good reproducibility, good recovery.</td>
</tr>
<tr>
<td></td>
<td>Campylobacter spp.</td>
<td>Sufficient</td>
<td>Large number of non detects and sufficient data to estimate PDF coefficients. PDF variance high. No recovery data.</td>
</tr>
<tr>
<td></td>
<td>Adenovirus adjusted</td>
<td>Good</td>
<td>Few non detects. Sufficient data to estimate PDF coefficients. Good recovery.</td>
</tr>
<tr>
<td></td>
<td>Rotavirus adjusted</td>
<td>Insufficient</td>
<td>Almost no detects. Data insufficiently credible to use.</td>
</tr>
<tr>
<td>Diurnal Variation</td>
<td>enterococci</td>
<td>Good</td>
<td>High frequency of good count data, good reproducibility, good recovery.</td>
</tr>
<tr>
<td></td>
<td>E. coli</td>
<td>Good</td>
<td>High frequency of good count data, good reproducibility, good recovery.</td>
</tr>
<tr>
<td></td>
<td>C. perfringens</td>
<td>Good</td>
<td>High frequency of good count data, good reproducibility.</td>
</tr>
<tr>
<td></td>
<td>FRNA Coliphage adjusted</td>
<td>Sufficient</td>
<td>Initial run (1) had poor detection and poor recovery. Data not used in PDF estimation.</td>
</tr>
<tr>
<td>Hydraulics</td>
<td>Flow</td>
<td>Good</td>
<td>Full set of high quality data. Flows consistent with one another.</td>
</tr>
<tr>
<td></td>
<td>Rainfall</td>
<td>Good</td>
<td>Full set of daily data.</td>
</tr>
</tbody>
</table>
Notes:

1. Following the Sydney Water Cryptosporidium crisis a review of the laboratories led to concerns about pathogen recovery rates. This led the Australian National Association of Testing Authorities (NATA) to undertake an evaluation of laboratory proficiency bearing in mind the difficult nature of pathogen enumeration. They proposed satisfactory pathogen recovery should be characterised by recovery rates in the range 10% to 110%. We have used this same range criterion to assess whether spike recovery was optimal or not.

2. Rating (Good, Sufficient, Insufficient) is assessed based on whether a) the data set is largely complete; and b) > 50% of ‘detects; and c); reproducibility of replicates within an order or magnitude; and d) recovery is good; and e) no other known problems or inconsistencies in data. The key index organisms were all from “Good” or ‘Sufficient’ data sets.

Pathogens and Enterococci from Weekly Monitoring

The weekly monitoring data was used to assemble a summary PDF set (Table 3) describing the source material water quality. Rotavirus was omitted as there were insufficient detection to generate a usable function. Its replacement with Adenovirus was considered reasonable as this group is a very common in faecal matter and sewage and is explicitly identified as a major cause of illness/outbreaks in bathing situations (World Health Organisation, 2006; World Health Organization, 2003).

Overall the data suggested that there was limited or no reduction in pathogen levels by the treatment. There was some accumulation in the WAS most notably in the case of Giardia. Giardia were the most abundant pathogen. Adenovirus were notable for the very small SD values.

Table 3. PDFs of pathogens from Weekly Sampling Data

<table>
<thead>
<tr>
<th>Waste Stream</th>
<th>Microorganism</th>
<th>Log10 average</th>
<th>Log10 SD</th>
<th>minimum</th>
<th>maximum</th>
<th>Count of detects</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Screened Primary</td>
<td>Cryptosporidium</td>
<td>1.203</td>
<td>0.31</td>
<td>BDL</td>
<td>1.7</td>
<td>4 of 12</td>
<td>oocysts/L</td>
</tr>
<tr>
<td>Screened Primary</td>
<td>Giardia</td>
<td>3.661</td>
<td>0.385</td>
<td>3.00</td>
<td>4.4</td>
<td>12 of 12</td>
<td>cysts/L</td>
</tr>
<tr>
<td>Screened Primary</td>
<td>enterococci</td>
<td>5.608</td>
<td>0.44</td>
<td>5.04</td>
<td>6.6</td>
<td>11 of 11</td>
<td>cfu/100mL</td>
</tr>
<tr>
<td>Screened Primary</td>
<td>Campylobacter spp.</td>
<td>-0.066</td>
<td>1.084</td>
<td>BDL</td>
<td>2.5</td>
<td>5 of 12</td>
<td>mpn/L</td>
</tr>
<tr>
<td>Screened Primary</td>
<td>Adenovirus</td>
<td>2.225</td>
<td>0.143</td>
<td>BDL</td>
<td>2.4</td>
<td>10 of 12</td>
<td>pfu/L</td>
</tr>
<tr>
<td>Secondary</td>
<td>Cryptosporidium</td>
<td>1.368</td>
<td>0.409</td>
<td>BDL</td>
<td>2.1</td>
<td>9 of 12</td>
<td>oocysts/L</td>
</tr>
<tr>
<td>Secondary</td>
<td>Giardia</td>
<td>2.23</td>
<td>0.508</td>
<td>1.43</td>
<td>3.2</td>
<td>12 of 12</td>
<td>cysts/L</td>
</tr>
<tr>
<td>Secondary</td>
<td>enterococci</td>
<td>5.352</td>
<td>0.317</td>
<td>4.86</td>
<td>6</td>
<td>11 of 12</td>
<td>cfu/100mL</td>
</tr>
<tr>
<td>Secondary</td>
<td>Campylobacter spp.</td>
<td>0.425</td>
<td>0.468</td>
<td>BDL</td>
<td>3.3</td>
<td>7 of 12</td>
<td>mpn/L</td>
</tr>
<tr>
<td>Secondary</td>
<td>Adenovirus</td>
<td>1.586</td>
<td>0.447</td>
<td>BDL</td>
<td>2.4</td>
<td>10 of 12</td>
<td>pfu/L</td>
</tr>
<tr>
<td>WAS</td>
<td>Cryptosporidium</td>
<td>2.412</td>
<td>0.139</td>
<td>BDL</td>
<td>2.8</td>
<td>4 of 12</td>
<td>oocysts/L</td>
</tr>
<tr>
<td>WAS</td>
<td>Giardia</td>
<td>4.55</td>
<td>0.337</td>
<td>4</td>
<td>5.1</td>
<td>12 of 12</td>
<td>cysts/L</td>
</tr>
<tr>
<td>WAS</td>
<td>enterococci</td>
<td>5.914</td>
<td>0.457</td>
<td>4.95</td>
<td>6.4</td>
<td>11 of 11</td>
<td>cfu/100mL</td>
</tr>
<tr>
<td>WAS</td>
<td>Campylobacter spp.</td>
<td>-0.2</td>
<td>1.108</td>
<td>BDL</td>
<td>1.8</td>
<td>4 of 12</td>
<td>mpn/L</td>
</tr>
<tr>
<td>WAS</td>
<td>Adenovirus</td>
<td>1.948</td>
<td>0.28</td>
<td>BDL</td>
<td>2.3</td>
<td>12 of 12</td>
<td>pfu/L</td>
</tr>
</tbody>
</table>

Notes:
1. BDL = Below Detection Limit


**Indicators**

As with pathogens a summary PDF set (Table 4) was constructed using the diurnal sampling data. Of the indicators FRNA were the most problematic. They showed high SDs and there was a strong suggestion that they were multiplying in the WAS.

**Table 4. PDFs for Indicators from Diurnal Sampling Data**

<table>
<thead>
<tr>
<th>Material</th>
<th>Microorganism</th>
<th>Log$_{10}$ average</th>
<th>Log$_{10}$ SD</th>
<th>minimum</th>
<th>maximum</th>
<th>Count of detects</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Screened Primary</td>
<td>enterococci</td>
<td>5.687</td>
<td>0.182</td>
<td>5.30</td>
<td>6</td>
<td>19 of 19</td>
<td>cfu/100mL</td>
</tr>
<tr>
<td>Screened Primary</td>
<td><em>E. coli</em></td>
<td>6.834</td>
<td>0.211</td>
<td>6.23</td>
<td>7.3</td>
<td>18 of 18</td>
<td>mpn/100mL</td>
</tr>
<tr>
<td>Screened Primary</td>
<td><em>C. perfringens</em></td>
<td>5.232</td>
<td>0.615</td>
<td>4.11</td>
<td>6.1</td>
<td>19 of 18</td>
<td>cfu/100mL</td>
</tr>
<tr>
<td>Screened Primary</td>
<td>FRNA Coliphage</td>
<td>2.853</td>
<td>0.422</td>
<td>BDL</td>
<td>3.5</td>
<td>9 of 18</td>
<td>pfu/L</td>
</tr>
<tr>
<td>Secondary</td>
<td>enterococci</td>
<td>5.18</td>
<td>0.38</td>
<td>4.43</td>
<td>5.7</td>
<td>19 of 19</td>
<td>cfu/100mL</td>
</tr>
<tr>
<td>Secondary</td>
<td><em>E. coli</em></td>
<td>6.434</td>
<td>0.253</td>
<td>5.98</td>
<td>6.8</td>
<td>18 of 18</td>
<td>mpn/100mL</td>
</tr>
<tr>
<td>Secondary</td>
<td><em>C. perfringens</em></td>
<td>5.076</td>
<td>0.542</td>
<td>3.60</td>
<td>6.1</td>
<td>18 of 18</td>
<td>cfu/100mL</td>
</tr>
<tr>
<td>Secondary</td>
<td>FRNA Coliphage</td>
<td>3.762</td>
<td>0.614</td>
<td>2.24</td>
<td>4.3</td>
<td>12 of 12</td>
<td>pfu/L</td>
</tr>
<tr>
<td>WAS</td>
<td>enterococci</td>
<td>6.028</td>
<td>0.285</td>
<td>5.30</td>
<td>6.4</td>
<td>18 of 18</td>
<td>cfu/100mL</td>
</tr>
<tr>
<td>WAS</td>
<td><em>E. coli</em></td>
<td>6.67</td>
<td>0.316</td>
<td>6</td>
<td>7.4</td>
<td>17 of 17</td>
<td>mpn/100mL</td>
</tr>
<tr>
<td>WAS</td>
<td><em>C. perfringens</em></td>
<td>5.849</td>
<td>0.285</td>
<td>5.17</td>
<td>6.3</td>
<td>17 of 17</td>
<td>cfu/100mL</td>
</tr>
<tr>
<td>WAS</td>
<td>FRNA Coliphage</td>
<td>4.998</td>
<td>0.2</td>
<td>4.69</td>
<td>5.2</td>
<td>11 of 12</td>
<td>pfu/L</td>
</tr>
<tr>
<td>Screened Primary</td>
<td>FRNA Coliphage</td>
<td>2.424</td>
<td>0.62</td>
<td>1.11</td>
<td>3.5</td>
<td>12 of 12</td>
<td>pfu/L</td>
</tr>
<tr>
<td>Screened Primary</td>
<td>FRNA Coliphage</td>
<td>1.955</td>
<td>1.247</td>
<td>0.619</td>
<td>3.5</td>
<td>18 of 18</td>
<td>pfu/L</td>
</tr>
<tr>
<td>Secondary</td>
<td>FRNA Coliphage</td>
<td>3.585</td>
<td>0.61</td>
<td>2.24</td>
<td>4.3</td>
<td>18 of 18</td>
<td>pfu/L</td>
</tr>
<tr>
<td>WAS</td>
<td>FRNA Coliphage</td>
<td>3.662</td>
<td>1.926</td>
<td>0.619</td>
<td>5.2</td>
<td>17 of 17</td>
<td>pfu/L</td>
</tr>
</tbody>
</table>

Notes:
1. The in FRNA coliphage data from diurnal run 1 were suspect. The PDF shown in normal fonts comes from a data set with these data removed. The PDFs from the whole data set are shown in italics and illustrate the resulting very high variance probably reflecting variations in method effectiveness.

**Microbial Quality Control**

Full Quality Control data results can be found in Appendix 23 Wastewater and WAS Quality.

**Replicated Samples**

In the case of the pathogen measurements the average ratio of replicate 1 to replicate 2 ranged from 0.55 to 1.48. In the case of indicator measurements the average ratio ranged from 0.5 to 1.58. These illustrate the limits of reproducibility of assays between samples collected concurrently. However the numbers are sufficiently close to indicate good replication sufficient for log transformed data.

**Replicated Diurnal Collection**
The three bacterial indicators were detected in similar and consistent levels. The FRNA Coliphage were not detected consistently in diurnal variation sample run 1. This was considered unlikely and so load estimation utilised only runs 2 and 3 (see above also).

**Collection Blanks**

Of 55 blanks for pathogens two failed in the case of Adenovirus. The level of one was comparable to that in the samples suggesting a mix up at the subcontracting laboratory.

In the case of the indicator study there was some contamination of the blank water containers. However the contamination was only of the order of 1 to 10 mpn or cfu per 100mL compared to counts of \(10^5\) to \(10^7\) per 100mL in actual samples. So it appears collection technique was satisfactory.

**Spikes**

Enterococci recovery was very good with an average of 27 compared to expect 30 cfu.100mL\(^{-1}\). Similarly \(E.\ coli\) spike counts were 36, 21, 28 compared to an expected value of 30. There was no indication here of poor recovery.

Recovery of Giardia on the other hand could be very poor and ranged from 0 to 41%. For each of the three waste streams a different recovery factor was estimated:

- **20±19%** (Primary);
- **5±4 %** (Secondary)’
- **3±1 %** (WAS)

Despite this cysts were frequently detected in the waste streams.

**Cryptosporidium** showed low recovery:

- **5.8±4.1 %** (Primary);
- **3.0±1.4%** (Secondary)’
- **8.3±6.7 %** (WAS)

Compared to **Giardia**, Cryptosporidium detection tended to be more sporadic. The suboptimal recovery of the protozoans indicated that the PDF estimates were somewhat uncertain and sensitivity analysis should be undertaken.

Poliovirus recovery was very good noting the spiked numbers were relatively high at \(ca\ 10^6\).  

- Primary recovery 69±12%
- WAS Recovery 69±4%
- Secondary recovery 72±3%

Poliovirus recovery was used to estimated Adenovirus recovery and for adjusting the counts, noting that the assay method used was a cell culture based one.

Only one of the three FRNA spike assays was unsuccessful. Recovery data from runs 2 and 3 were used to adjust the counts of these indicator bacteriophage.

**Carryover**

Carryover occurring in the autosampler appeared to be low and comparable to that observed in the blanks collected concurrently. Number of indicators in blanks were typically < 50 per 100mL compared to sample levels >10\(^4\). 100mL\(^{-1}\) so the use of the autosampler was seen as reasonable and the data used.
**Microbial Population Variance**

Microbial numbers in wastewater have been reported to undergo seasonal variation (Payment *et al.*, 2001).

Examination of the collected in this study showed however:

1. No clear trend over the course of the study in the daily pathogen measurements with the possible except of *Giardia* in WAS (Figure 2);
2. No trend in the indicator counts over the course of each day during the diurnal variation study (Figure 3);
3. Little relationship between flow and pathogen numbers (Figure 4).

These data suggested together that the 9am measurements of pathogen levels were representative of WWTP effluent quality at other times and hence the level PDFs could be used in QMRA.
c.

\[ y = -31.549x + 1 \times 10^6 \quad R^2 = 0.0135 \]

\[ y = -823.2x + 3 \times 10^7 \quad R^2 = 0.3369 \]

\[ y = 2.919x - 115729 \quad R^2 = 0.0352 \]

d.
e. Figure 2. Timeseries plots of pathogen levels from weekly measurements

Notes:
1. a. = enterococci ; b. = Cryptosporidium ; c. = Giardia ; d. = Campylobacter ; e. = Adenovirus