Lake Parramatta, 2015. Photograph by Chris Frape.

# Strategic Analysis of Water Quality in the Parramatta River

Parramatta River Catchment Group

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## How should recreational water quality in the Parramatta River be assessed? A Review of Current Literature.

Parramatta City Council | FINAL

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# Contents

# Executive Summary

A review of current scientific literature was undertaken with the following key aims:

- To identify current knowledge regarding water quality in the Parramatta River with direct relevance to potential increased primary contact recreational activities such as swimming.
- To consider potential future approaches to monitoring recreational water quality in the Parramatta River to assess the public health safety for potential increased primary contact recreational uses.

The Parramatta River has a long history of contamination, primarily related to urban and industrial activities which have taken place in the river's catchment throughout the last two centuries. Research conducted since the 1990s has established a relatively clear picture of spatial and temporal patterns of contamination for a range of chemical contaminants including metals, nutrients, halogenated organic compounds and polycyclic aromatic hydrocarbons.

Information relating to cyanobacteria and pathogenic microbial contaminants (bacteria, viruses and protozoa) are much sparser. Some bacterial monitoring (for faecal indicator bacteria) has been undertaken at a limited number of sites over an extensive period of time. While this monitoring has been undertaken with the aim of providing an indication of likely faecal contamination –and hence the presences of a potentially wider range of organisms – there are limitations to the ability to draw such connections.

A detailed understanding of recreational water quality risks in the Parramatta River will require comprehensive risk assessment. Suitable established frameworks are identified to undertake the necessary risk assessment for chemical contaminants. While a number of limitations are identified for current approaches to water quality monitoring for microbial risks in Australian recreational waters, a general approach to developing a suitable strategy is proposed.

However, it is concluded that the establishment of an effective ongoing monitoring strategy will require efforts to understand the observable relationships between potential indicators of faecal microbial contamination and risks associated with the actual presence of pathogenic organisms. Quantitative Microbial Risk Assessment (QMRA) is identified as a tool which can be used to facilitate the development of this understanding.

It is proposed that a number of contaminants and indicators should be monitored at specific locations proposed for future recreational water use. Due to the nature of the contaminants of concern, and whether they present acute or chronic health risks, there are significant differences in appropriate monitoring frequencies. Furthermore, it is essential to consider the appropriate sites for sample collection, including whether they include sediment, the pelagic zone or the water surface.

Monitoring data should be used in association with a conceptual recreational exposure assessment to identify levels of chemical and microbial health risk posed to recreational water users at individual proposed sites.

Note that these recommendations follow a thorough review of the scientific literature and are considered to provide a scientifically-informed assessment of recreational water quality. However, this process has not included a cost/benefit analysis since that is to be undertaken separately.

## Heavy metals in surface sediment

A small suite of heavy metals (most notably mercury, cadmium and lead) should be assessed in sediment at each proposed swimming location. Since these contaminants present chronic (as opposed to acute) human health risks, monitoring could be done well in advance of any recreational use and updated periodically (eg. Once per year at the beginning of each swimming season).

## Dioxins in surface sediment and aqueous samples

Key dioxins that have previously been detected in significant concentrations in the Parramatta River should be assessed in each proposed swimming location. Analytical methods that target dioxins concentrations commonly target a suite of these chemicals, enabling a weighted sum of the results to give a toxic equivalency

(TEQ) value, representing the overall toxicity as a single value. The TEQ should be measured for sediment and aqueous samples on an annual basis at the start of each swimming season.

## Cyanobacteria

The presence of severe cyanobacterial blooms can be detected visually and regular (eg, weekly) inspections should be made at various locations on the river. A clear protocol for detecting and identifying a potential cyanobacterial bloom should be developed. In circumstances where a bloom is detected as affecting, or potentially affecting a swimming site, a number of actions could be taken. One option may be to close the site for recreational use as a precautionary measure. Alternatively, samples could be genetically analysed for specific species in order to determine whether they are potentially toxin-producing.

## Enterococci

As the current international standard for monitoring recreational water quality in marine waters, regular monitoring (during the swimming season) for enterococci is warranted. Although significant background concentrations can be anticipated, and the usefulness as an indicator of human faecal contamination is limited, monitoring will strengthen the overall evidence-base and ability to understand changing water quality. A once-off period of intensive monitoring (eg, daily for 3-6 months) would provide a strong basis for understanding trends. It would then be anticipated the ongoing regular testing could be less frequent and possibly event-driven.

## Bacteroides

Monitoring of *Bacteroides* is undertaken using molecular methods (as opposed to culture-based methods). It has the advantages that these bacteria are excreted in very high numbers and that a range of host-specific organisms provides excellent opportunities to distinguish various sources of bacterial contamination in water. Human-specific *Bacteroides* should be tested, potentially along with other strains which may indicate sources from birds and other species. Analysis should be undertaken with the same frequency as analysis for enterococci.

## Faecal bacteriophages

Faecal bacteriophages (such as coliphages) should be tested since they are believed to represent the fate of enteric viruses, much more effectively than bacterial monitoring. An appropriate faecal bacteriophage species (and hence method) will need further consideration. Analysis should be undertaken with the same frequency as analysis for enterococci.

## Direct pathogen monitoring

Some direct pathogen monitoring, for non-faecally derived pathogenic organisms, such as pathogenic *Vibrio Spp.* may be warranted. Following a satisfactory research period, it may be possible to develop sufficient understanding of the spatial and temporal variability, that other non-microbial measures may become sufficient to inform risk assessments for these organisms.

## Spill detection

Careful consideration should be given for how best to detect a diverse range of spills that may occur and pose risks to recreational water quality. Some water quality monitoring (e.g. for BTEX chemicals) may be effective. However, other useful approaches could include visual inspections, combined with a compulsory reporting-based regulatory system, or a well organised community reporting system.

# 1. Introduction

The Parramatta River extends from Blacktown Creek in the west to the confluence of the Lane Cove River in the east. It is the largest river entering Port Jackson (Sydney Harbour). The river is tidal to the Charles Street Weir in Parramatta, some 30 km upstream from Sydney Heads. The Parramatta River below the weir is, in fact, an estuary and not a river, whereas (Upper) Parramatta River above the weir is a river.

The Parramatta River is 19 km in length, but has around 220 km of waterways in its catchment, including a number of significant tributaries (Parramatta River Catchment Group, 2016). These include Subiaco Creek, Tarban Creek, Duck River, Duck Creek, Haslams Creek, Iron Cove Creek, Hawthorne Canal and Powells Creek.

The estuary itself covers 12 square kilometres and is in a constant state of flux with tidal movements and freshwater from the river's tributaries changing the chemical composition of the water on a daily basis (Parramatta River Catchment Group, 2016). Tidal flushing for complete water exchange takes 3-4 months.

The total area of the catchment is 257 $km^2$ and the majority is managed by local government, including the Local Government Areas of Ashfield, Auburn, Bankstown, Blacktown, Burwood, Canada Bay, The Hills, Hunters Hill, Holroyd, Leichhardt, Parramatta, Ryde and Strathfield. Other major land managers include Sydney Olympic Park Authority, Bidjigal Reserve and Shell Oil Clyde Refinery (Parramatta River Catchment Group, 2016).

Catchment management oversight is provided by the Parramatta River Catchment Group (PRCG), which is composed of Local Government and other agencies (PRCG 2016). The current management strategy for the river is outlined in The Parramatta River Estuary Coastal Zone Management Plan, which was prepared on behalf of the PRCG (Cardno Pty Ltd 2013). The Sydney Coastal Councils Group have also recently reported the outcomes of a Sydney Harbour Coastal Zone Management Plan Scoping Study (GHD, 2015)

The Parramatta River and the Sydney estuary, to which it drains, have a long history of anthropogenic contamination by chemical and microbial substances, stretching back to European colonisation in 1788 (Davies & Wright, 2014; Birch *et al.*, 2015c). Together, they form one of the most modified waterways in Australia due to a highly urbanised catchment and a high population (Birch *et al.*, 2015c).

Industries were first established on the banks of Darling Harbour in 1800, from where they gradually spread along the southern shore of the River (Birch *et al.*, 2015b). While regulatory reforms were introduced in the 1940s and 1950s to address pollution of the Harbour, it wasn't until the Clean Waters Act 1970 that pollution levels entering the Harbour began to decline (Montoya, 2015).

Many parts of the estuary have been subject to land reclamation, and in many cases these reclaimed lands have been filled with municipal or other waste materials (Birch *et al.*, 2009). This reclamation has been shown to have detrimentally impacted water quality in the Parramatta River and its tributaries in a number of circumstances (Suh *et al.*, 2003; Suh *et al.*, 2004). Industrial activities throughout the 20th century, along the shores and within the catchment of the River (Birch *et al.*, 2015c), have resulted in a large store of legacy chemicals in sediments (Birch & Taylor, 1999; Birch & Taylor, 2000).

However, the greatest contemporary source of contaminants is believed to be stormwater, particularly from highly urbanised catchments on the southern side of the river (Beck & Birch, 2012). Evidence for this includes a strong spatial relationship between major stormwater outlets and distribution of metals in Sydney Harbour sediment (Birch & McCready, 2009; Birch & Rochford, 2010). Stormwater modelling reported in 2009 indicated an average annual discharge to Sydney Harbour of 215 GL (Birch & Rochford, 2010), much of which is delivered to the lower reaches of the estuary via the Parramatta River.

Urban stormwater often conveys a high load of human pathogens (disease-causing microorganisms) including viruses, bacteria and protozoa (Jiang *et al.*, 2015; Lim *et al.*, 2015; Page *et al.*, 2015). Many of these pathogens originate from municipal sewage and are transferred from sewers to stormwater systems by leakage or wet-weather overflows designed into the sewage system (Passerat *et al.*, 2011; Khan *et al.*, 2014). Consequently, increased rainfall, runoff and stormwater overflow lead to more events carrying peak concentrations of waterborne pathogens in surface water (Schijven & de Roda Husman, 2005; Schijven *et al.*, 2013). This can lead to the exposure of recreational water users to pathogens responsible for a range of illnesses, including gastrointestinal disease, skin infections and ear infections (Stewart *et al.*, 2008).

A review of water quality data collected for the upper Parramatta River between 1990 and 2007 was commissioned by Parramatta Council (Laxton *et al.*, 2008). This review reported a number of indicators of generally poor water quality including elevated nutrient concentrations, turbidity, and –following wet weather – faecal coliforms. Salinity concentrations exhibited a large range from 0.06 ppt to 31ppt, depending on runoff and tidal movements. Large seasonal variations were observed in surface temperature and dissolved oxygen concentrations. While the ecology of the upper-Parramatta River was found to be very productive and dynamic, the water quality failed to meet most of the relevant criteria for primary contact recreation, secondary contact recreation, or passive recreation.

A number of sites on the lower Parramatta River are currently monitored for recreational water quality, most notably Cabarita Beach (Cabarita), Chiswick Baths (Five Dock Bay) and the Dawn Fraser Pool (Balmain). During 2014-2015, these sites were assessed as having water quality that was safe for swimming most of the time but could be susceptible to pollution after heavy rainfall (10 mm or more) (OEH NSW 2015). However, the river west of Cabarita is not currently considered to be swimmable due to poor water quality, including the assumed presence of pathogenic microorganisms.

Community consultation undertaken in preparation of The Parramatta River Estuary Coastal Zone Management Plan highlighted "*water quality suitable for recreational usage*" as a contemporary community value (Cardno Pty Ltd 2013). Consequently, a key management aim identified in the Plan is "*to improve water quality in the estuary such that it is suitable for a range of environmental functions and recreational uses*" (Cardno Pty Ltd 2013).

As a first step toward an improved understanding of recreational water quality, a review of existing published literature pertaining to chemical and microbial contaminants within the river has been undertaken. It is intended that this review will provide a basis for informed discussion and future planning.

# 2. Chemical Contaminants

Research undertaken since the 1990s has established a relatively detailed knowledge base for the spatial and temporal distribution of a range of chemical contaminants in the Parramatta River. Most attention has been focused on metals, nutrients, halogenated organic compounds and polycyclic aromatic hydrocarbons. More recently, a survey of water soluble trace organic contaminants was published, highlighting the continuing flux of contaminants to the river from stormwater and sewage systems that leak or overflow to stormwater.

## 2.1 Metals and metalloids

Sediments in Sydney Harbour, including the Parramatta River and its tributaries are contaminated with a range of inorganic substances, including heavy metals and metalloids (Birch & Taylor, 1999; McCready *et al.*, 2006a; Ying *et al.*, 2009; Birch, 2011). Widely detected metals and metalloids include antimony, arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, silver, and zinc (McCready *et al.*, 2006a). In particular, severe enrichments (above historic concentrations) of copper, lead and zinc have been reported (Irvine & Birch, 1998; Nath *et al.*, 2014).

Many of the metals detected in sediment are also measurable as dissolved (or suspended) substances in the water column, including cadmium, copper, nickel, manganese and zinc (Hatje *et al.*, 2001; Hatje *et al.*, 2003). The partitioning of metals between precipitated and dissolved forms depends on local geochemical factors including pH and redox conditions (Simpson *et al.*, 2002). Sediment contamination around Iron Cove was found to be greatest near a stormwater canal, where sediments were anoxic and contained high concentrations of sulfide in the pore-water (Simpson *et al.*, 2002).

Since metals concentrations (copper, lead and zinc) in the water column have been found to be associated with total suspended solids (TSS) concentrations, monitoring TSS has been proposed as a surrogate for estimating metal loading in real time under some conditions (Beck & Birch, 2011). Research has shown that TSS and associated metal loads are elevated in Parramatta River water column during heavy rainfall and high-wind conditions (Birch & O'Hea, 2007).

Surficial sediments covering the bathymetry of the parts of the Parramatta River contain some of the highest concentrations of metals reported in Australia, and globally (Birch *et al.*, 2013). Of six embayments of Sydney Harbour recently studied, sedimentary metal concentrations were highest in Rozelle and Blackwattle Bays, followed closely by Iron Cove, for both total and size-normalised sediments (Birch *et al.*, 2013). Lower concentrations were observed in Hen and Chicken Bay, Homebush Bay, Lane Cove estuary and Central Middle Harbour (with the latter two external to the Parramatta River estuary).

Sediment cores taken adjacent to long-term industrial sites have shown that past industrial practices contributed significantly to contamination of estuarine sediment (Taylor *et al.*, 2004; Birch *et al.*, 2015b). Further analysis of vintage surficial sediment and sediment cores has revealed that metal concentrations in surficial sediment have declined in many areas since about the early 1990s (Birch *et al.*, 2013). The decline was reported to have been extensive in the upper and central parts of the estuary, but accompanied by smaller increases in the lower estuary, due mainly to a down-estuary shift in industry and urbanisation.

Mangrove systems, and the fine sediments that provide their root rhizosphere, play an important role influencing the fate of trace metals in the Parramatta River. Although they have a large potential to sequester trace metals, thus reducing pollution in the marine environment, changes in physico-chemical conditions may trigger release of accumulated trace metals to the sediment–water interface (Chaudhuri *et al.*, 2014). The ability to detect heavy metal concentrations in mangrove pneumatophore (aerial root) tissues has led to suggestions that these may play a useful role as a bio-indicator of estuarine metal contamination (Nath *et al.*, 2014).

The majority (>90%) of contemporary metal (copper, lead and zinc) and total suspended solid annual loads are believed to be contributed during high stormwater-flow conditions (>50 mm rainfall day) (Beck & Birch, 2012). There is evidence to indicate that commercial and industrial discharges to the stormwater system are current major contributors to overall metal discharged to the river and that significant improvements could be achieved by targeting these sources (Beck & Birch, 2014). A further minor source of metals is from surface runoff from previously contaminated sites on the river foreshore (Birch *et al.*, 2013). In particular, soil concentrations of copper, lead and zinc are known to be elevated throughout large areas of the previously industrialised catchment (Snowdon & Birch, 2004; Birch *et al.*, 2010b). As such, managing catchment condition to prevent particulate transport from sources such as soils, road dust and drainage canals may provide a further effective control on the quality of riverine sediment (Birch & McCready, 2009).

## 2.2 Halogenated organic compounds and polycyclic aromatic hydrocarbons

Parramatta River sediments are contaminated in many areas with organochlorine compounds (Birch & Taylor, 2000). Some of the most prominent organochlorine contaminants include pesticides (chlordane, DDT/DDD/DDE, Aldrin, dieldrin, endrin, heptachlor, heptachlor epoxide and hexachlorbenzene) and polychlorinated biphenyls (PCBs) (Birch & Taylor, 2000; McCready *et al.*, 2006a; Ying *et al.*, 2009).

Analytical assessment of the onset of organochlorine compound contamination of estuarine sediment closely agrees with the record of industrialisation in the catchment of each embayment (Taylor *et al.*, 2004). PCBs, chlordane, and to a lesser extent dieldrin, are most elevated in sediment in creeks on the southern shores of the estuary, suggesting sources within older, highly urbanised/industrialised catchments of western-central Sydney (Birch & Taylor, 2000). There are high concentrations of total DDT and hexachlorbenzene in sediments of the upper harbour and Homebush Bay suggesting that (previous) chemical industries on the shores of the estuary in this area are sources of these contaminants (Birch & Taylor, 2000).

Most notoriously, a range of dioxin compounds are known to contaminate sediment throughout the Parramatta River, with the most intense contamination in and around Homebush Bay (Mueller *et al.*, 2004; Birch *et al.*, 2007; Ying *et al.*, 2009). The term ‘dioxins’ is used to refer to two main groups of polychlorinated hydrocarbons. These are polychlorinated dibenzo para dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). Certain dioxin-like PCBs with similar toxic properties are also commonly included under the term ‘dioxins’.

Between 1928 and 1985, Timbrol (later known as Union Carbide) manufactured a variety of chemicals on the Rhodes Peninsula. Dioxins were released in industrial effluent disposed of in part in Homebush Bay (Montoya, 2015). An Environmental Impact Statement prepared prior to remediation of the Timbrol site and parts of Homebush Bay reported the detection of some of the highest surface and subsurface sediment dioxin concentrations in the world, as well as significant aqueous concentrations (Parsons Brinckerhoff Australia & Thiess Serivces, 2002). A distinct congener profile corresponded to the chemicals known to have been produced at the Timbrol site (Birch *et al.*, 2007). The Timbrol site and adjacent Homebush Bay sediments were partially remediated between 2005 and 2011, but as of 2014, the effectiveness of the remediation program was unknown (Montoya, 2015).

In addition to the high sediment loads of dioxins, passive samplers have been used to assess dioxin concentrations in the pelagic zone of the river (Roach *et al.*, 2009). Highest concentrations were measured in Homebush Bay and concentrations generally declined with distance from Homebush Bay. Higher daily concentrations were detected in summer compared to winter, which may reflect higher rates of solubilisation in warmer water.

Polybrominated diphenyl ethers (PBDEs) and hexabromocyclododecane (HBCDs) have been used extensively as brominated flame retardants worldwide. Stormwater is a recognised potential source of brominated flame retardants to the aquatic environment (Richman *et al.*, 2013). These hydrophobic compounds are resistant to many common biodegradation processes, rendering them likely to accumulate in estuarine sediment (Richman *et al.*, 2013).

A range of PBDEs and HBCDs have been reported in sediment cores from the Sydney estuary, including sites on the Parramatta River, taken from locations close to stormwater drains (Drage *et al.*, 2015). Large increases in concentrations were observed for all compounds between 1980 and 2014 (Drage *et al.*, 2015). PBDE concentrations in surficial sediments were relatively high when compared with those presented in the available international literature, suggesting that their input into the estuary has not decreased since their bans almost a decade earlier. HBCD concentrations exhibited a sharp increase during the 1990s and with global legislation allowing its usage for the next 10 years, it is expected that its input into the estuary is likely to continue.

Perfluorinated compounds including perfluorooctane sulphonate (PFOS) and perfluorooctanoate (PFOA) are persistent environmental pollutants which have become of concern worldwide in recent years following their detection in the general population of several countries, and in the global environment (Chang *et al.*, 2014;

MiraIles-Marco & Harrad, 2015; Wang *et al.*, 2015). PFOS and PFOA have been reported in water and surface sediment samples collected from around Homebush Bay, in the upper reaches of the Parramatta River estuary (Thompson *et al.*, 2011; Kaserzon *et al.*, 2012).

The river sediments also contain a range of polycyclic aromatic hydrocarbons (PAHs) including acenaphthene, acenaphthylene, anthracene, benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(ghi)perylene, benzo(a)pyrene, benzo(e)pyrene, coronene, chrysene, dibenz(ah)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, 2-methylnaphthalene, naphthalene, perylene, phenanthrene, and pyrene (McCready *et al.*, 2000; McCready *et al.*, 2006a). The spatial distribution of PAHs indicates that they derive predominantly from urban run-off (McCready *et al.*, 2000). The relative abundance of individual PAH compounds indicates that high temperature combustion processes are the predominant source of PAH contamination (McCready *et al.*, 2000; Ying *et al.*, 2009).

## 2.3 Nutrients

Nutrients, including nitrogen and phosphorus are transported to the river by urban stormwater (Beck & Birch, 2012). Modelling nutrient loads from stormwater to the Parramatta River (using MUSIC) has indicated that the majority is delivered during moderate rainfall (5–50 mm day$^{-1}$) conditions and accumulates close to discharge points and remains in the estuary (Birch *et al.*, 2010a). In contrast to metals, considerable loads of nitrogen (up to 55%) and phosphorus (up to 21%) appear to be transported to the River during dry weather base-flow conditions (Beck & Birch, 2012).

Under high-rainfall conditions (>50 mm day$^{-1}$), the estuary becomes stratified and nutrients are either removed directly in a plume or indirectly by advective/dispersive remobilisation (Birch *et al.*, 2010a).

Nutrients have accumulated in bottom sediments in concentrations up to 50 times greater than pre-anthropogenic levels (Birch *et al.*, 1999).

## 2.4 Water soluble trace organic contaminants

A survey of trace organic contaminants in various locations of the Parramatta River was recently reported (Birch *et al.*, 2015a). The contaminants detected included pharmaceuticals, personal care products, a food additive (acesulfame) and pesticides. Anthropogenic organic chemicals such the pharmaceuticals and acesulfame are increasingly used as indicators of domestic sewage since there are usually no other significant sources to the environment (Gasser *et al.*, 2014; Tran *et al.*, 2014; Jekel *et al.*, 2015; Lee *et al.*, 2015a). Since there are no sewage treatment plants discharging to the Parramatta River, the most viable sources for these chemicals are leakage from sewerage systems and occasional sewage overflows to the stormwater system during intense wet weather events.

This survey reported that acesulfame and a number of the pharmaceuticals were widely detected, even in the absence of significant recent rainfall (Birch *et al.*, 2015a). This is suggestive of a continuous source from leaking sewers. The major source for a number of pesticides detected was proposed to be after environmental application and wash-off to stormwater (Birch *et al.*, 2015a). However, it should be noted that most of the reported pesticides are also known to occur in municipal sewage (Ratola *et al.*, 2012; Pal *et al.*, 2014; Margot *et al.*, 2015).

## 2.5 Surfactants

Municipal sewage typically contains high concentrations of a range of surfactants including linear alkylbenzene sulfonate, quaternary ammonium compounds, alkylphenol ethoxylates, and alcohol ethoxylates (Jardak *et al,* 2016). Although many of these surfactants are relatively biodegradable, high concentrations in sewage can lead to relatively high concentrations in surface waters (Jardak *et al,* 2016). At sufficiently high concentrations, this may present environmental risks including aquatic toxicity and foaming, which may lead to reduced oxygen concentrations (Jardak *et al,* 2016).

# 3. Microbial Contaminants and Indicators

In contrast to chemical contaminants, microbial water quality contaminants are generally regarded as posing greater risks to human health than to aquatic ecosystems. Important microbial contaminants which may present health risks to recreational water users include bacteria, viruses, protozoa and cyanobacteria.

Throughout the last century, the Parramatta River has generally been regarded as a contaminated waterway and this has resulted in minimal use involving primary recreational contact (eg, swimming). As a consequence, microbial monitoring has been minimal, other than that undertaken by the NSW Office of Environment and Heritage (OEH) under the Beachwatch program (Hose *et al.*, 2005).

## 3.1 Bacteria

The NSW OEH has routinely monitored faecal contamination at a number of Sydney Harbour sites as part of the Beachwatch program since 1994 (Hose *et al.*, 2005). This monitoring is based on a single water sample collected from each site by boat every six days during summer season (October to April) and monthly surveillance sampling undertaken during the winter period (May to September). Samples are analysed for enterococci using culture-based methods following the Australian standard method (Standards Australia & Standards New Zealand, 2007). Three of the monitored sites are on the Parramatta River at Cabarita Beach (Cabarita), Chiswick Baths (Five Dock Bay) and the Dawn Fraser Pool (Balmain).

Spatial and rainfall related patterns of faecal coliform and enterococci densities in Sydney Harbour estuary have been reported (Hose *et al.*, 2005). Unsurprisingly, sites to the west and along the Parramatta River exhibited much greater bacterial densities than eastern sites, closer to the mouth of the Harbour. This was attributed to greater tidal flushing at sites closer to the harbour mouth. At particular locations, it was determined that rainfall accounted for between 15-66% of the observed temporal variability in bacterial densities, with the strongest relationships for the Parramatta River sites. The findings of this research indicate that simple rainfall-based regression models are appropriate for predicting bacterial indicator concentrations when flushing at a site is limited.

*Vibrio* are a genus of naturally occurring marine bacteria that have substantial environmental and human health importance. A number of sites within the Parramatta River were recently investigated for the presence of *Vibrio Spp*. (Siboni *et al.*, 2016). Significant spatial and seasonal variation were observed with abundance higher during late summer than winter and within locations with mid-range salinity (5–26 ppt). While toxigenic strains of *V. cholerae* were not detected in any samples, non-toxigenic strains were detected in 71% of samples. In contrast, pathogenic *V. vulnificus* was only detected in 14% of samples, with its occurrence restricted to the late summer.

High water temperature is a strong predictor for the presence of *Vibrio spp.* and they are mainly detected in warmer waters (above 15°C) (Lutz *et al.*, 2013; Vezzulli *et al.*, 2013). Many studies have demonstrated that the abundance of *Vibrio spp.* follows a seasonal pattern, largely dictated by temperature (Lutz *et al.*, 2013; Rashid *et al.*, 2013). It is believed that warmer temperatures enhance the persistence of *Vibrio spp.* by promoting biofilm formation and colonisation of environmental surfaces such as chitin (Stauder *et al.*, 2010). The effect of increased ocean surface temperatures in promoting spread of *Vibrio spp.* in coastal and brackish waters has been considered as a possible causal factor explaining observed increases in *Vibrio* illnesses in many parts of the world (Vezzulli *et al.*, 2013).

Several QMRAs have suggested that bacteria, -most notably Campylobacter - are significant sources of health risk from some urban rivers and freshwater lakes (Oster *et al.*, 2014; Corsi *et al.*, 2016).

## 3.2 Viruses

No published reports of virus densities in the Parramatta River have been identified. However, pathogenic human viruses are believed to cause over half of gastroenteritis cases associated with recreational water use worldwide (Ahmed *et al.*, 2015a).

## 3.3 Protozoa

Cryptosporidium and Giardia are intestinal protozoan parasites, which are excreted by infected hosts in large numbers, hence are commonly present in untreated sewage (Li *et al.*, 2012; Tonani *et al.*, 2013; Taran-Benshoshan *et al.*, 2015). Furthermore, these parasites are environmentally robust, which facilitates transmission (Cacciò *et al.*, 2005). In circumstances where recreational marine waters are contaminated with domestic sewage, there may be considerable risk of infection and disease caused by these organisms (Betancourt *et al.*, 2014).

No published reports of protozoan densities in the Parramatta River have been identified.

## 3.4 Cyanobacteria

No published reports of cyanobacterial populations in the Parramatta River have been identified. Nonetheless, it is known that cyanobacteria blooms do occur in the upper reaches of the river from time-to-time and this should be considered for primary recreational contact.

# 4. Recreational Water Quality Risks

Australian guidelines for managing risks in recreational water identify a diverse range of physical, chemical and biological hazards to which swimmers may be exposed (NHMRC 2008). These include water depth, currents, UV radiation, water temperature, snags, venomous species and other dangerous animals. Each of these would need to be considered for a comprehensive risk assessment of future potential recreational water use. However, in the current circumstance, only chemical and microbial water quality contaminants are discussed.

## 4.1 Risks from recreational exposure to chemical contaminants

Many of the contaminants reported in sediments and the water column of the Parramatta River may pose toxicity risks to the aquatic system and biota (Birch & Taylor, 2002a; Birch & Taylor, 2002b; McCready *et al.*, 2004; Birch *et al.*, 2008). Indeed, correlations between measures of ecological toxicity and concentrations of some chemical contaminants in surficial sediments have been observed (McCready *et al.*, 2005; McCready *et al.*, 2006b).

Furthermore, many heavy metals and lipophilic organic chemicals (such as dioxins and many pesticides) are known to accumulate in fish and other aquatic organisms (Hellou *et al.*, 2013; Lehnherr, 2014; Cruz *et al.*, 2015). Examples of this accumulation or 'biomagnification' have been reported for various chemicals in the Parramatta River. These include metals (Birch *et al.*, 2014; Lee *et al.*, 2015b), chlorinated hydrocarbons (Roach & Runcie, 1998), brominated hydrocarbons (Losada *et al.*, 2009), and perfluorinated hydrocarbons (Thompson *et al.*, 2011). As a consequence of biomagnification of dioxins in various fish species, the NSW Government have imposed a ban commercial fishing in Sydney Harbour and advise against the consumption of fish caught by recreational fishing west of the Sydney Harbour Bridge (NSW DPI 2016).

In addition to the movement of industry from the upper-catchment, metal concentration declines have been partially attributed to the introduction of regulation, which prevents pollutants being discharged directly to the river (Birch *et al.*, 2013). Relevant legislation have included the State Pollution Control Commission Act 1970, the CleanWaters Act 1978, the Coastal Protection Act 1979 and the Catchment Management Act 1989, which have prohibited the dumping of waste, reformed pollution licensing and enforced control procedures (Smith, 1997). Nonetheless, since high stormwater metal concentrations persist, it is proposed that stormwater remediation will be required for further rapid improvement (Birch *et al.*, 2013).

Cyanobacterial-derived water quality impairment issues are a growing concern worldwide, including health risks from recreational water exposure (Otten & Paerl, 2015). These organisms can produce a range of bioactive metabolites, many of which are known human toxins. The occurrence and characterisation of cyanobacterial organisms –and the metabolites that they may exude- appears to be a current knowledge gap for the Parramatta River.

In the late 1990s, Sydney Water undertook a risk assessment to assess the risks of chemical contaminants to human health (and aquatic organisms) in creeks, rivers, estuaries and ocean waters affected by wet weather discharges (Bickford *et al.*, 1999). This work included assessment of chemicals in stormwater and sewage overflows (as well as sewage treatment plant discharges). The risk assessment methodology was based on comparison of measured and predicted concentrations of chemicals with toxicity reference values. While various risks to human health were identified (primarily from the consumption of contaminated fish), no significant risks from chemicals to people engaged in recreational water use were identified.

## 4.2 Risks from recreational exposure to microbial contaminants

Recreational water use commonly involves exposure to pathogenic organisms and is associated with increased levels of illnesses including gastrointestinal illness respiratory illness, as well as ear, eye and skin infections (Collier *et al.*, 2015). Epidemiological studies have shown that risks of these infections in swimmers increases with either exposure to urban runoff or declining water quality due to pollution or sewage (Stewart *et al.*, 2008). Based on risk assessments, it has been estimated that globally, each year, there are in excess of 120 million cases of gastrointestinal disease and in excess of 50 million cases of more severe respiratory diseases caused by swimming and bathing in wastewater-polluted coastal waters (Shuval, 2003).

Despite the apparent lack of historic monitoring data, there is compelling evidence that a variety of pathogenic microorganisms may present some risks to people engaging in primary recreational contact with Parramatta River water. This is because the chemical monitoring data (see previous section), undertaken over decades, has confirmed that stormwater is a major source of pollution to the river. Stormwater is a well-established source of pathogen risk to recreational water environments (Stewart *et al.*, 2008).

Following a large storm, large volumes of stormwater flow into the upper Parramatta River and cause significant stratification (Wolanski, 1977). Such circumstances could lead to significant exposure to pathogens by swimmers. Recent investigations have revealed that these stratified stormwater plumes are relatively short-lived and generally not effective for rapidly transporting contaminants outside the Sydney estuary (Lee *et al.*, 2011). Thus the major mechanisms for recovery of microbial water quality will be dilution and gradual environmental inactivation of pathogens. The frequent stratification of the river greatly affects the fate and transport of contaminants and pathogens in the water column as well as the ability to assess the associated risks.

Numerical modelling has been employed to determine stormwater runoff volumes and establish an appropriate rainfall/runoff relationship capable of replicating fresh-water discharge due to the full range of precipitation conditions in the Parramatta River (Lee & Birch, 2012). International research has demonstrated that even relatively small drains can lead to localised high levels of faecal indicator bacteria at enclosed beaches (Rippy *et al.*, 2014).

In addition to fresh inputs of contaminants to the river, recreational users may be exposed to contaminants that are resuspended from the bottom sediment to the water column by a variety of processes. These include natural processes such as wind, waves or tidal movements, or anthropogenic processes including the effect of watercraft (Bishop, 2007).

Recreational activity itself can lead to increased concentrations of some contaminants including human waterborne pathogens, and faecal indicator bacteria in marine recreational beach water. For example, it is believed that re-suspension of bottom sediments by bathers may cause elevated levels of enterococci and waterborne parasites (Graczyk *et al.*, 2010). Swimmers may themselves be sources of microorganisms in water including enterococci and *Staphylococcus aureus* (Elmir *et al.*, 2007; Stewart *et al.*, 2008). Consequently, human pathogens can be present in beach water on days deemed acceptable for bathing according to faecal bacterial standards (Graczyk *et al.*, 2010).

A number of naturally occurring Vibrio spp. are pathogenic to humans (Baker-Austin *et al.*, 2013; Matteucci *et al.*, 2015). Consequently, an understanding of the spatiotemporal dynamics of *Vibrios* and their potential to bloom and cause disease outbreaks has become increasingly important (Oberbeckmann *et al.*, 2011; Takemura *et al.*, 2014). The spatiotemporal dynamics of *Vibrio Spp.* and the occurrence of bloom events have been linked to several environmental drivers, including temperature, salinity, turbidity, dissolved oxygen, pH, chlorophyll, and nutrients, as well as associations with potential host organisms (Takemura *et al.*, 2014).

# 5. Future Water Quality Risks and Management

Prior to declaring new sites on the Parramatta River as being safe for swimming (including under any specific limited circumstances), a number of risk assessment and ongoing monitoring activities should be undertaken. Principal guidance for this risk assessment should come from the Australian guidelines for managing risks in recreational water (NHMRC 2008). However, due to the historically degraded water quality of the river, and its vulnerability to a wide range of contaminant sources, a number of additional factors should be considered.

Many chemical contaminants pose predominantly chronic risks, as opposed to acute risks, to the health of recreational water users. Examples include heavy metals and halogenated organic chemicals such as dioxins (Nau, 2006; Bensefa-Colas *et al.*, 2011). An advantage for the assessment of these types of risks is that short-term exposure variations are much less significant than long-term exposure trends (enHealth Council, 2012). This means that effective risk assessment could be undertaken well in advance of potential recreational contact and need only be updated periodically (eg, annually).

Based on the information collected in this review, site-specific risk assessment should be undertaken for exposure to a range of chronic risk chemicals including heavy metals, halogenated organic compounds and polycyclic aromatic hydrocarbons.

In 1983, the US National Research Council published what became known as the 'red book' (NRC 1983), which provided a formalised set of steps to be taken for assessing risks to human health by chemicals from environmental and other sources. These were:

1. Problem Formulation and Hazard Identification—to describe the human health effects derived from any particular hazard (for example, acute toxicity, carcinogenicity)
2. Exposure Assessment—to determine the size and characteristics of the population exposed and the route, amount, and duration of exposure
3. Dose-Response Assessment—to characterise the relationship between the dose exposure and the incidence of identified health impacts
4. Risk Characterisation—to integrate the information from exposure, dose response, and health interventions in order to estimate the magnitude of the public health impact and to evaluate variability and uncertainty.

These steps have evolved into a general framework now used by environmental health agencies throughout the world to assess risks posed by environmental human health hazards. An Australian version is described in detail in the important document *Environmental Health Risk Assessment: Guidelines for Assessing Human Health Risks from Environmental Hazards* (enHealth Council, 2012). The systematic approach described in this document is appropriate for the assessment of chronic exposure risks to chemical contaminants in the Parramatta River.

Some potential chemical contaminants to the Parramatta River may be associated with possible short-term extreme exposure. Examples include those resulting from large chemical spills (from within the catchment, direct industrial releases, or from ships). Such circumstances could lead to acute health risks to recreational water users. Similarly, cyanobacterial growth could lead to rapid production and release of chemicals posing health risks to swimmers. These types of acute health risks, resulting from short-term exposure may be overlooked when focussing risk assessment on the enHealth Council (2012) framework. This is because that document does not explicitly emphasise the need to consider potential 'hazardous events' which may drastically alter short-term exposure conditions. For a more systematic consideration of the role of hazardous events in risk assessment, Australian guidelines for managing risks in recreational water (NHMRC 2008), or even the Australian Drinking Water Guidelines (NHMRC & NRMMC 2011) should be referred to.

Pathogenic microbial organisms (including viruses, bacteria and protozoa) present predominantly acute risks to public health. Consequently, short-term fluctuations in pathogen densities can be highly significant in terms of changing health risks to recreational water users.

Management approaches for recreational water quality currently focus on sampling recreational waters and returning a water quality statistic possessing many uncertainties, days after swimmer exposure has occurred or averaging samples collected over multiple dates to generally characterise a bathing site (Ashbolt *et al.*, 2010).

The NSW Government Beachwatch program (OEH NSW 2015) grades swimming sites as Very Good, Good, Fair, Poor or Very Poor in accordance with the Australian Guidelines for Managing Risks in Recreational Water (NHMRC 2008). These grades provide a long-term assessment of how suitable a beach is for swimming. The grades are determined from the most recent 100 water quality results (two to four years' worth of data depending on the sampling frequency) and a risk assessment of potential pollution sources. Water quality results are based primarily on enterococci monitoring. The risk assessment is based on sanitary surveys considering potential pathogen sources including bathers, animals, toilet facilities, sewage overflows, sewer chokes, stormwater discharges and other local features (OEH NSW 2015). Beachwatch also issues daily beach pollution forecasts for swimming sites in Sydney, Central Coast, Hunter and Illawarra regions. The likelihood of pollution is predicted using rainfall data. Rainfall thresholds have been determined for each swimming location by analysing rainfall and bacterial data collected over the last five years. Managers at some swimming sites, including Dawn Fraser Pool, use this information to close the baths when pollution is likely.

Scientific evidence supporting recreational water quality benchmarks primarily stems from epidemiological studies conducted at beaches impacted by human faecal sources. The human illness potential from a recreational exposure to freshwater impacted by rainfall-induced runoff containing non-human animal faecal material may be at least an order of magnitude lower than the benchmark level of public health protection associated with current US recreational water quality criteria, which are based on contamination from human sewage sources (Soller *et al.*, 2014; Soller *et al.*, 2015).

Nonetheless, a range of human pathogenic (and indicator) microorganisms have been detected in the faeces of domestic animals and wildlife in the Sydney region (Cox *et al.*, 2005). Of particular relevance to the Parramatta River may be birds, some species of which are known to be sources of pathogenic bacteria including Salmonella and Campylobacter (Antilles *et al.*, 2015; Cody *et al.*, 2015; Ramonaite *et al.*, 2015).

Current technology for water-quality monitoring is based on frequent sampling and culturing methods which are time-consuming and do not allow a rapid decision making process (Amini & Kraatz, 2015). However, during the last few decades, a range of 'molecular methods' such as polymerase chain reaction (PCR)-based methods, DNA sequencing and immuno-fluorescence methods have been developed, and offer promising strategies for more rapid and cost-effective pathogen detection (Amini & Kraatz, 2015).

Compared to culture-based methods, which can take between 18 and 96 hours for sample processing, molecular methods can be applied in just a couple of hours (Raith *et al.*, 2014). Thus, molecular methods offer an opportunity to provide a more meaningful statement of microbial risk to water-users by providing near-real-time information enabling potentially more informed decision-making with regard to recreational water use (Mendes Silva & Domingues, 2015; Oliver *et al.*, 2016).

Furthermore, the potential use of a suite of molecular methods would facilitate monitoring of some indigenous organisms, which are not directly associated with faecal indicators, such as pathogenic *Vibrio Spp.* (Siboni *et al.*, 2016).

However, the use of molecular methods, compared with traditional culture-based methods for quantifying microbial parameters in recreational waters generates considerable ongoing debate at the science-policy interface (Oliver *et al.*, 2016). Regulators and researchers have highlighted a number of technical and logistical issues associated with the emerging use of molecular methods for recreational water quality assessment (Oliver *et al.*, 2014). For example, the practical benefit of rapid analysis can be limited by centralised laboratory infrastructure, implying long transit times for some locations. There also remain uncertainties regarding the robustness of the epidemiological evidence-base for results obtained from molecular method quantitation (Oliver *et al.*, 2014).

For comparing faecal indicator monitoring data with acceptable guideline levels, some agencies have chosen the 95th percentile (WHO 2003; NHMRC 2008). However, it is well known that these kinds of data do not display a normal distribution and several alternative distributional forms have been proposed and are in use for estimating the percentile (Laura Patat *et al.*, 2015).

To facilitate and standardise the process, a Microsoft Excel template (the EnteroTester) has been developed for generating workbooks that estimate the infection risk (according to formula used in the above guidelines) for any given enterococcal distribution, and calculate a 95th percentile standardised to that of the reference distribution with the same risk (Lugg *et al.*, 2012). A similar statistical decision support tool 'EnterosisA' was recently developed to facilitate the analysis of microbial water quality data for the purposes of classifying recreational waterways in south-east Queensland (Xie *et al.*, 2015). Other approaches, such as the use of a Tweedie distribution have also been proposed (Laura Patat *et al.*, 2015).

An alternative (or complimentary) approach for the rapid assessment of water quality may be achieved by the development of an understanding of the relationships between pathogen (or indicator) organism densities and rapidly observable factors that influence their occurrence. It is this type of observation that has led to recommendations to avoid swimming at Sydney beaches for one to three days following rainfall events (OEH NSW 2015).

In a recent study from Germany, multiple linear regression models were developed in order to predict the abundance of the faecal indicator organisms *E. coli*, intestinal enterococci and somatic coliphages in a freshwater river (Herrig *et al.*, 2015). Useful predictive variables were found to include ammonia concentration, turbidity, global solar irradiance, chlorophyll a content, rainfall, oxygen concentration and pH. It was reported that these models could explain around 70% of the observed variance in faecal indicator concentrations. Site-specific efficiencies were around 80%.

Predictive empirical modelling has been used at a number of Great Lakes beaches in Chicago to develop a fully automated water quality management system (Shively *et al.*, 2016). In that study, data from water quality buoys and weather stations were transmitted to a web hosting service. An executable program simultaneously retrieved and aggregated data for regression equations and calculated empirical modelling results each morning. In that case, solar radiation, rainfall, and wind direction were common model regressors. A decision-making tool for beach management has also been described, based on (up to 5-day) advance predictions of bathing water quality, which is in-turn derived from weather forecasts (Bedri *et al.*, 2016).

A range of calculation and modelling techniques are available for estimating annual stormwater contaminant loadings to the Parramatta River. These include calculations based on measured event mean concentrations (EMCs), pollutant load/runoff relationship calculations and the use of the computer modelling application ‘Model of Urban Stormwater Infrastructure Conceptualisation’ (MUSIC). However, a recent assessment of the use of these techniques to estimate annual TSS, TN, TP and metals loadings to the Parramatta River revealed high degrees of inconsistencies among predictions (Beck & Birch, 2013). Consequently, it was concluded that some or all of the methods used were likely to be inaccurate and unreliable for this application.

Quantitative Microbial Risk Assessment (QMRA) has been an increasingly important tool for assessing risks from recreational water exposure to pathogenic microbial organisms (Oster *et al.*, 2014; Fewtrell & Kay, 2015; Soller *et al.*, 2015; Sunger & Haas, 2015). The use of QMRA for modelling (and thus predicting) pathogens and indicators has numerous potential advantages (Ashbolt *et al.*, 2010). For example, it can facilitate the exploration of a large diversity of scenarios for faecal contamination and hydrologic events, such as from waterfowl, agricultural animals, resuspended sediments and from the bathers themselves (Ashbolt *et al.*, 2010).

Epidemiological studies show a generally elevated risk of gastrointestinal illness in bathers compared to non-bathers but often no clear association with water quality as measured by faecal indicator bacteria; this is especially true where study sites are impacted by non-point source pollution (Fewtrell & Kay, 2015). Evidence from QMRAs support the lack of a consistent water quality association for non-point source-impacted beaches. It has been suggested that source attribution, through quantified microbial source apportionment, linked with appropriate use of microbial source tracking methods should be employed as an integral part of future epidemiological surveys (Fewtrell & Kay, 2015).

## 5.1 Indications of faecal contamination and associated pathogens

Most contemporary recreational water quality monitoring practices are focused on concentrations of faecal indicator bacteria such as *total and faecal coliforms, Clostridium perfringens, Escherichia coli, and faecal enterococci* (Field & Samadpour, 2007; Bennell *et al.*, 2015; Cheung *et al.*, 2015; Fan *et al.*, 2015).

However, monitoring for these indicators is not always effective for determining when streams and coastal waters are contaminated with sewage because faecal indicator bacteria can reside in the environment and may even multiply under certain conditions (Field & Samadpour, 2007; Stewart *et al.*, 2008; Boehm *et al.*, 2009a). For example, these indicators have been consistently reported from beach sands and sediments from freshwater and marine beaches, even in the absence of any known sources of human/animal waste (Stewart *et al.*, 2008; Cloutier *et al.*, 2015; Halliday *et al.*, 2015). Since these "extra-enteric" bacteria multiplied in environmental habitats, such as soil rather than intestinal habitats of humans or animals, these bacteria are not indicators of faecal contamination (Fujioka *et al.*, 2015).

In some cases, constant background levels of faecal indicator bacteria are present due to the presence of birds and wildlife. This was the case for Lake Parramatta, for which background *E. coli* and enterococci contamination were detected due to waterfowl (Roser *et al.*, 2006).

Furthermore, faecal indicator bacteria are known to differ in their environmental fate and transport characteristics compared to some important pathogens. For example, indicator bacteria have been shown to be more sensitive to inactivation by sunlight than some important waterborne human viruses (Fujioka & Yoneyama, 2002; Boehm *et al.*, 2009b; Carratala *et al.*, 2013). Consequently, the concentrations of faecal indicator bacteria measured in environmental waters are not always reliable –or even conservative- assurances that human pathogens are below levels that may cause infections in swimmers (Field & Samadpour, 2007; Fujioka *et al.*, 2015).

Consequently, there are limitations to determining risks to recreational water users based only on monitoring data for established faecal indicator bacteria. Promising more sewage-specific markers have been identified including *C. perfringens*, various bacteriophages, *Bacteroides*, as well as human enteric viruses (Boehm *et al.*, 2009a; Fujioka *et al.*, 2015). Ecology and physiology of indicator bacteria varies vastly from those of many pathogens, such as viruses and protozoa. Therefore, there is a need for supplemental indicator organisms that would be indicative of risk for a wide array of human pathogens and to provide better protection of public health (Fujioka *et al.*, 2015).

### 5.1.1 E. Coli

*Escherichia coli* (*E. coli*), is a member of the coliform group of bacteria, but unlike the general coliform group, *E. coli* are almost exclusively of faecal origin and their presence is thus an effective confirmation of faecal contamination. However, *E. coli* are not specific to humans and are excreted by many species including birds. Most strains of *E. coli* are harmless, but some can cause serious illness in humans.

*E. coli* is used as a faecal indicator for fresh water (but not marine) systems in the USA as described by the US EPA Recreational Water Quality Criteria (US EPA 2012). In that document, local authorities are provided two alternative recommended levels of health protection that they may work to. Recommendation 1 is applied at an estimated illness rate of 36/1000 and is based on exceeding *E. coli* densities with a geometric mean of 126 cfu/100 mL or statistical threshold value of 410 cfu/100 mL. Recommendation 2 is applied at an estimated illness rate of 32/1000 and is based on exceeding *E. coli* densities with a geometric mean of 100 cfu/100 mL or statistical threshold value of 320 cfu/100 mL. The statistical threshold value approximates the 90th percentile of the water quality distribution and is intended to be a value that should not be exceeded by more than 10 percent of the samples taken.

### 5.1.2 Enterococci

Enterococci are facultative anaerobic organisms (that is, they are capable of cellular respiration in both oxic and anoxic environments). Though they are not capable of forming spores, enterococci are tolerant of a wide range of environmental conditions including temperature (10-45°C), pH (4.5-10.0) and high sodium chloride concentrations (Fisher & Phillips, 2009).

Enterococci are excreted in faeces and are rarely present in unpolluted waters. International agencies including the US EPA (US EPA 2012) and the World Health Organization (WHO 2003) now generally use enterococci as the single preferred faecal indicator in marine waters.

Australian guidelines for recreational water quality monitoring do not dictate the use of enterococci monitoring (NHMRC 2008). However, they do refer explicitly to WHO recommendations for the use of enterococci as a faecal indicator in marine waters (WHO 2003).

The WHO Guidelines describe an assessment approach based on a combination of a sanitary inspection (to identify susceptibility to faecal influence) and microbial water quality assessment (WHO 2003). The microbial water quality assessment is used to categorise recreational water quality based on measurements of the 95$^{th}$ percentile intestinal enterococci densities. The applied microbial water quality assessment categories are “A” (≤40 cfu/100mL), “B” (41-200 cfu/100mL), “C” (201-500 cfu/100 mL) and “D” (>500 cfu/100 mL).

The WHO Guidelines also refer to 95th percentile value of intestinal enterococci/100ml greater than 500 (or greater than 200 if source mainly human faecal pollution) in consecutive samples as one of a number of conditions that may merit intervention by public health authorities (WHO 2003).

Analogous to the use of *E. coli* for fresh water quality criteria, the US EPA provides two recommendations for enterococci limits in marine and fresh waters, relating to estimated illness rates of 36/1000 (Recommendation 1) and 32/1000 (Recommendation 2) (US EPA 2012). Recommendation 1 is based on exceeding enterococci densities with a geometric mean of 35 cfu/100 mL or statistical threshold value of 130 cfu/100 mL. Recommendation 2 is based on exceeding enterococci densities with a geometric mean of 30 cfu/100 mL or statistical threshold value of 110 cfu/100 mL.

### 5.1.3 Clostridium perfringens

*Clostridium perfringens* is an anaerobic, spore-forming pathogenic bacterium. *C. perfringens* is ever-present in nature and can be found as a normal component of decaying vegetation, soil, marine sediment, and in the intestinal tract of humans and other species.

In Hawaii, due to ambient growth and high elevated concentrations of faecal indicator bacteria in soil and streams, *C. perfringens* is used as a secondary water quality standard because it is a more reliable marker of sewage contamination than the conventional faecal indicator bacteria (Viau *et al.*, 2011b; Fujioka *et al.*, 2015). This practice was adopted because monitoring data had shown that the ambient and daily concentrations of conventional faecal indicator bacteria in the major streams of Oahu, Hawaii greatly exceeded the (now defunct) faecal coliform standard (Fujioka *et al.*, 1988) and the US EPA *E. coli* and enterococci standards (Luther & Fujioka, 2004). Widespread occurrence and growth of the conventional faecal indicator bacteria in the soil environment of Hawaii was determined to be the reason for high levels of these bacteria in freshwater streams and rivers (Fujioka *et al.*, 2015). Consequently, it was not possible to determine when streams and coastal waters were contaminated with sewage or other human faecal sources.

In contrast to conventional faecal indicator bacteria, concentrations of *C. perfringens* were consistently low in streams but increased during a sewage contamination event (Fujioka *et al.*, 2015). As a result, the state of Hawaii adopted *C. perfringens* as a state recreational water quality standard (HIDOH 2014). However, the US EPA questioned the use of *C. perfringens* for this purpose and recommended that Hawaii maintain enterococci as the primary faecal indicator and use *C. perfringens* as a secondary tracer (Fujioka *et al.*, 2015).

### 5.1.4 Bacteroides

*Bacteroides* is a genus of obligate anaerobic bacteria, making up a substantial portion of the mammalian gastrointestinal flora. As many as $10^{10}$ cells per gram of human faeces (dry weight) have been reported (Franks *et al.*, 1998).

*Bacteroides* species have a high degree of host specificity that reflects differences in the digestive system of the host animal (Bernhard & Field, 2000). The use of molecular methods, instead of culture methods, to measure *Bacteroides* densities allows quantification of genetic markers that are specific to the host of the bacteria (Gómez-Doñate *et al.*, 2015). In this way, it is possible to distinguish between multiple bacterial sources, such as by human-*Bacteroides*, pig-*Bacteroides*, and bovine-*Bacteroides* (Viau *et al.*, 2011a). A number of specific *Bacteroides* assays are now available to identify faecal sources such as human, cattle, dogs and other animals (Fujioka *et al.*, 2015).

Since *Bacteroides* have a small potential to grow in the environment, they have been increasingly touted as an alternative or complimentary sewage-specific faecal indicator bacteria (Bell *et al.*, 2009; Ahmed *et al.*, 2015b; Byappanahalli *et al.*, 2015; Verhougstraete *et al.*, 2015). Epidemiological studies have demonstrated that human-specific *Bacetroides* densities correlate well with illness rates, supporting the use of these bacteria as reliable indicators of human health risks in recreational waters (Wade *et al.*, 2006).

### 5.1.5 Bacteriophages

A bacteriophage is a virus that infects and replicates within a bacterium. “Coliphage” is used to describe bacteriophage that infect coliform bacteria, such as *E. coli* (Eg., Bacteriophage lambda and Leviviridae). Another common term is faecal bacteriophage, to describe bacteriophage known to occur in faecal material (Ashbolt *et al.*, 2010).

Coliphages have been studied as pollution indicators since the early 1960s (Fujioka *et al.*, 2015). Municipal sewage contains high concentrations of bacteriophages and, since they multiply in specific bacterial strains found in sewage, monitoring recreational waters for coliphages is an alternative indicator for sewage pollution (Ashbolt *et al.*, 2010). More recently, other bacteriophages , such as those that infect enterococci (enterophages) and those that infect *Bacteroide*s bacteria, have been considered as alternative sewage-specific indicators (Fujioka *et al.*, 2015).

Since bacteriophages closely resemble human viruses (but are not infective to human cells), they are conceptually better models of human enteric virus behaviour in the environment than faecal indicator bacteria (Ashbolt *et al.*, 2010). This is significant since enteric viruses are commonly thought to be the major causative agents for observed recreator gastroenteritis in waters impacted by human sources (Sinclair *et al.*, 2009; Schoen & Ashbolt, 2010; Soller *et al.*, 2010).

### 5.1.6 Direct monitoring of pathogens

Given that presence and prevalence of indicators does not often correlate well with risk of infection from environmental pathogens, direct monitoring for pathogens is often suggested. Detection of pathogens from environmental samples is increasingly viable through the development of molecular methods such as PCR. While the detection of pathogen molecular material is not necessarily indicative of viability or even infectivity, it does represent a rapid and specific method. New molecular assays have been introduced for detection of bacterial, viral and protozoan pathogens (Stewart *et al.*, 2008). Furthermore, recent improvements in detection technologies are allowing simultaneous detection of multiple targets in a single assay from the same sample (Stewart *et al.*, 2008).

Quantitative assessment of bacterial pathogens was recently reported from Great Lakes beaches in North America (Oster *et al.*, 2014). Quantitative PCR was used to test for genes from E. coli O157:H7, shiga-toxin producing *E. coli*, *Campylobacter jejuni*, *Shigella spp*., and a *Salmonella enterica*-specific DNA sequence in algae, water, and sediment. Subsequent QMRA, based on the detected pathogen densities indicated a moderate probability of illness for *Campylobacter jejuni* at the sites investigated.

Methicillin-resistant Staphylococcus aureus (MRSA) have also been detected in fresh and marine recreational water bodies (Fogarty *et al.*, 2015). Since this and other non-faecal derived bacteria have been observed not to be related to faecal indicator presence, direct monitoring of recreational waters for non-faecal bacteria such as staphylococci and/or Staphylococcus aureus has been suggested (Stewart *et al.*, 2008; Fogarty *et al.*, 2015).

Following this logic, it has been proposed that a range of microbial measurements could also be applied as indicators of disease transmitted via other routes of infection, including respiratory routes (e.g. adenovirus, Legionella) and oral ingestion or wound infections from indigenous microbes such as *Vibrio Spp.* (Stewart *et al.*, 2008).

Direct monitoring of human pathogenic viruses in recreational waters has shown that the presence of these pathogens may be poorly represented by faecal indicator bacteria (Prez *et al.*, 2015; Updyke *et al.*, 2015). Consequently, enteric viruses have been widely recommended (but yet to be successfully implemented on a large scale) in routine monitoring of water quality (Prez *et al.*, 2015; Updyke *et al.*, 2015).

Despite the advances in detection technologies, a major limitation for direct pathogen monitoring has been the low concentrations in environmental waters (Ahmed *et al.*, 2015a). For appropriately sensitive analysis, it is necessary to effectively concentrate and recover pathogens such as viruses from large volumes of water (eg. >10L) (Ahmed *et al.*, 2015a). Many of the reported methods to concentrate pathogens are complex, costly and/or of low efficiency (Fujioka *et al.*, 2015).

# 6. Conclusions

The Parramatta River is an iconic waterway in Australia's largest city. An opportunity to improve water quality suitable for (primary contact) recreational usage appears to have growing community and other stakeholder support. However, there are a number of water quality issues that require improved understanding and risk management before widespread recreational use of some parts of the river can be advised.

A variety of chemical contaminants are known to be present in suspended and surface sediment, as well as in the pelagic zone. While previous assessments have not identified significant risks to recreational water users from chemical contaminants, such risks should be carefully re-examined on a site-by-site basis. Human health risk assessments from exposure to known contaminants such as heavy metals, halogenated organic compounds, polycyclic aromatic hydrocarbons and surfactants should be undertaken at specific sites proposed for recreational activities. Australian environmental health risk assessment guidelines (enHealth Council, 2012) provide an appropriate framework for this assessment.

Additional consideration should be given to the potential for hazardous events, such as spills, to lead to short-term elevated exposure of chemicals which may present acute health risks. Australian guidelines for managing risks in recreational water (NHMRC 2008) and the Australian Drinking Water Guidelines (NHMRC & NRMMC 2011) provide an appropriate framework for this type of risk assessment.

Pathogens, such as viruses, bacteria and protozoa have a number of potential sources in the river. The most significant of these is expected to be stormwater, which is commonly contaminated with raw (untreated) municipal sewage. Since these organisms present predominantly acute health risks, short-term water quality fluctuations can be highly significant in terms of the risks they present to human health. Indigenous pathogenic bacteria, including *Vibrio Spp*. may also present health risks to recreational water users, yet the spatial and temporal variability in their occurrence is currently only poorly understood.

Existing approaches in Australia for microbial monitoring in recreational water are based almost exclusively on monitoring the faecal indicator bacteria enterococci. However, there are a number of limitations with this approach. These include the occurrence of organisms from sources other human faeces or sewage. Indeed, constant background detections of *E. coli* and enterococci (believed to have originated from waterfowl) were reported from risk assessment work undertaken for Lake Parramatta (Roser *et al.*, 2006). A similar outcome might be expected for the Parramatta River. Pathogenic *Vibrio Spp.* have also been shown to be indigenous to the river (Siboni *et al.*, 2016). A further limitation of faecal indicator bacteria monitoring is that they are not believed to be conservative indicators for some of the most important sewage-derived pathogens including a number of enteric viruses (Field & Samadpour, 2007; Fujioka *et al.*, 2015).

Internationally, a tiered monitoring strategy has been proposed as potentially viable option for future recreational water quality (Boehm *et al.*, 2003; Stewart *et al.*, 2008). In a tiered approach, the first step could involve using the simplest and most practical tests for contamination, perhaps using a rapid molecular test for faecal indicators. Tier two may involve adding methods to differentiate human from animal sources of pollution, and the tier three test, if necessary, would measure for specific pathogens.

A similar approach proposed for future review of US Recreational Water Quality Criteria may be worth investigating (Fujioka *et al.*, 2015). This involves routine monitoring for culturable levels of *C. perfringens* (considered to be the most conservative assay for sewage-borne bacteria) and for human specific *Bacteroides* (highly sensitive and specific assay for human faecal contamination). A water sample with minimal *C. perfringens* density would be interpreted as having minimal sewage contamination. However, since *C. perfringens* is not specific to human sewage, elevated concentrations could indicate either human or animal faecal contamination. Therefore, by also assaying the water sample for human specific *Bacteroides*, it can be determined whether the source of *C. perfringens* was human. Additional *Bacteroides* assays can be used to determine the specific animal faecal sources.

Prior to the development of an on-going routine monitoring program for the Parramatta River, it will be necessary to characterise the observable relationships between potential indicators of faecal microbial contamination and the actual presence of pathogenic organisms. QMRA can be used to facilitate an understanding of risks associated with specific pathogenic substances and a wide range of circumstances including the occurrence of hazardous events which may lead to elevated level of exposure to pathogens.

Monitoring, especially for pathogens and organic contaminants, is expensive and interpreting results can be complex. While comprehensive monitoring is required in the short-term to understand spatial and temporal patterns of contamination, longer-term solutions may benefit from a greater focus on predictive modelling.

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