water e-journal
Online Journal of the Australian Water Association
ISSN 2206-1991
Volume 3 No 1 2018
https://doi.org/10.21139/wej.2018.011

EVALUATING THE SUITABILITY OF WATER QUALITY INDICES FOR THE HEALTH OF URBAN WATERWAYS

A CASE STUDY OF THE PARRAMATTA RIVER

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ABSTRACT

An appropriate water quality index is important for developing strategies for public health and wellbeing in a given urban area. This study set out to compare the National Sanitation Foundation Water Quality Index (NWQI) and Canadian Council of Ministers for the Environment Water Quality Index (CWQI) in terms of accuracy and overall utility, in the context of a parameter-limited water quality monitoring and management. To do this, water quality data were collected from the Upper Parramatta River, Sydney, Australia, a waterway draining a highly urbanised catchment. The parameters included in this study were dissolved oxygen, pH, electrical conductivity, total dissolved solids, turbidity, phosphates (as filterable reactive phosphates) and nitrates. Biological parameters were also tested, with macro-invertebrate surveys and bacteriological tests for faecal coliforms, total coliforms, Escherichia Coli (*E. coli*) and enterococci. Additionally, corridor imperviousness of tributaries was determined using GRASS-GIS and compared to index results. The results of this study showed that the NWQI did not act as an accurate representative of water quality, but did generally behave in line with current models of urban stream syndrome. The CWQI was found to be more accurate in terms of range but the modified CWQI based on some changes in the CWQI parameters was found to be more accurate. While this study cannot endorse the use of any index studied as a tool for meaningful water quality analysis, it does suggest that a modified CWQI be considered as a tool for the preliminary assessment of water quality and for community education in urban areas.

Keywords: Water quality monitoring, water quality index, urban stream, river health, and public health.

INTRODUCTION

One of the most consistently challenging aspects of scientific study lies in presenting results in a manner that can be understood by those outside your area of expertise. This becomes even more important in studies with potential public health impacts, as the results must then be presented in a manner understandable by the layperson. In studies and disciplines where numerous parameters influence the overall findings this is often accomplished through the consolidation of data into a single value, or index. Such is the case in water quality. Water quality is a general overarching term used to describe the physical, chemical, biological, aesthetic and radiological characteristics of water (NSW OEH, 2015).

These characteristics are often made up of a variety of parameters, with physical, chemical and biological parameters often being represented by a variety of different measurements.

The chemical component of water quality is one of the broadest, as based on its intended use and the thoroughness of testing. It can include an enormous number of parameters, representing selected heavy metal contaminants, levels of arsenic and other metalloids, as well as more site-specific chemicals when applicable. Despite this wide range, many studies choose to focus on the more general measurements of chemical contamination, including nutrient levels, pH and hardness. However, the chemical parameter included in the widest range of studies is dissolved oxygen, due to its well-established status as a water quality indicator (Kannel et al., 2007, Sanchez et al., 2007, Rudolf et al., 2002). On the other hand, biological components of water quality include measures of indicator bacteria such as faecal coliforms and *E. coli* (ANZECC, 2000a) amongst others. The bacteria tested, and the regime within which they are tested, are designed to match the intended use of the water. For example, when testing water for drinking, objective values are much more stringent than for recreational use.

Based on such tests, their component parameters and objectives, water may then be determined to be suitable or unsuitable for a particular use. As an example, for water to be considered fit for drinking purposes, it must perform well against biological characteristics such as bacteria, and chemical characteristics such as heavy metal content (ANZECC, 2000a). In contrast, for agricultural activities such as irrigation, water tests will focus on chemical characteristics (most notably salinity). Additionally, water quality can be an important factor in the protection of ecosystems, both aquatic and terrestrial (NSW OEH, 2015). As such, it is important that water quality be properly maintained in areas outside of basic human use. However, maintaining water quality can be difficult, due to its tendency to react strongly to various environmental factors, such as local geology, and the nature of the water source in question. One of the most influential of these factors, and the factor over which human behaviour has the most control, is land use.

URBAN STREAM SYNDROME

It has long been established that local land use can exercise a great deal of influence on local water quality and aquatic ecosystems. A key example of this is what Paul & Meyer (2001) referred to as 'Urban stream syndrome,' whereby increased urbanisation leads to changes in local water quality and hydrology.

A well-established effect of urban stream syndrome is its tendency to create a flashy hydrography (Paul & Meyer, 2001, Klein, 1979, Walsh et al., 2005). When a waterway has a flashy hydrography, its flow behaviour is characterised by low flow rates during dry periods, and extremely high flows after rainfall. This effect is also seen in changes to waterway topography, with channels becoming more highly incised due to increasingly violent stormwater flows.

The physiochemical effects of urban stream syndrome are well documented, with higher electrical conductivity linked to catchment imperviousness (Tippler et al., 2012). The behaviour of turbidity is also known to change in areas affected by Urban stream syndrome, with the characteristic behaviour of low turbidity that occasionally increases during rainfall events. The chemical effects of urban stream syndrome do undergo slight variations based on the local environment. Some of the common effects include greater variability in dissolved oxygen levels, often characterised by a night-time decrease (Paul & Meyer, 2001, Walsh et al., 2005); an increase in nutrient levels, notably phosphates and nitrates, that are exacerbated by rainfall; changes to pH; and increased toxicant levels.

Biologically, urban stream syndrome has been linked to decreases in both the biodiversity of macro-invertebrate assemblages and the prevalence of more sensitive species, (Collier & Clements, 2011, Davies et al., 2010, Tippler et al., 2012) as well as potential increases in algal biomass (Walsh et al., 2005).

Whilet no solid link currently exists between urban stream syndrome and bacteriological indicators, the disturbance of soil may result in the release of sedimentary bacteriological populations, which Alm et al. (2003) found in wet sand around freshwater beaches. Should that occur, then the released bacteria coupled with released nutrients may cause a sharp increase in local bacterial populations.

The main mechanism by which urban stream syndrome is known to act is as a result of catchment imperviousness (Paul & Meyer, 2001, Tippler et al., 2012, Collier & Clements, 2011, Walsh et al., 2005). Catchment imperviousness represents the proportion of the catchment that water cannot pass through and is often extremely high in urban areas. Additionally, only a small change in catchment imperviousness is required to produce discernible results, with Tippler et al. (2012) finding that a catchment imperviousness level of 5% correlated with an observable change in water quality and macro-invertebrate communities.

Stormwater infrastructure may also contribute to urban stream syndrome. First, in regards to the drain and pipe systems that are found around Sydney, work by Wright et al. (2011) found that the erosion of concrete infrastructure could be considered to be a contributing factor, before suggesting improved sealing of pipes to prevent said erosion. Second, the combination of stormwater and sewage overflows that occur across Sydney increase the potential for extremely poor quality water to be released as the result of a major rainfall event with potential impacts to human health.

However, despite the extensive research and knowledge about the various causes and impacts of Urban stream syndrome, communicating this information to the public and those in the planning and construction sector is difficult. This is due in part to the wide variety of effects associated with various land-use factors. As such, in this case, it may be necessary to consolidate data into as simple a form as possible. One example of how this is done through the use of a water quality index. Considering the importance of water quality indices in management of waterways, this study aimed to determine the relative accuracy and utility of indices in the context of a parameter-limited water quality monitoring.

WATER QUALITY INDICES

A water quality index is a mathematical tool used to simplify water quality reporting and comprehension through consolidating various markers of water quality into a single 0-100 index number (CCME, 2006).

Many attempts to create an accurate water quality index have been made. The two indices that stand out as the most used, both within studies and as the basis of other indices, are the National Sanitation Foundation Water Quality Index (NWQI) and the Canadian Council of Ministers for the Environment Water Quality Index (CWQI). Whilet both of these indices are widely used, flaws within them have prevented either from gaining universal acceptance. To understand the reasons for this, as well as examine and understand previous applications of the indices, it is necessary to review the relevant literature surrounding them.

NWQI

The NWQI was developed by Brown et al. in 1970. Generally based on the model proposed by Horton (1965) the concept was to update the parameters used, their relative weightings, and the Q_i value given to the specific parameters. In doing so, a panel of 142 water quality experts were sent three questionnaires (Wills & Irvine, 1996, Varnosfaderany et al., 2009). Using the first two of these questionnaires, nine parameters were chosen to be included in the index, those parameters being dissolved oxygen, faecal coliforms, biochemical oxygen demand, pH, turbidity, temperature change, nitrates, total phosphates and total solids. The third survey asked respondents, based on their professional opinion, to draw their best estimate of a sub-index for each parameter. These results were then averaged into the sub-index scores (Wills & Irvine, 1996, Varnosfaderany et al., 2009), and weighting values were given to each parameter based upon the results of statistical analysis. The resulting index is defined by the following equation:

$$NWQI = \frac{\sum_{i=1}^{n} Q_i \, W_i}{\sum_{i=1}^{n} W_i}$$

Where

Q_i = the sub-index value of the parameter i, and
W_i = the weighting value of the parameter i.

When using the full complement of parameters, only the first half of the equation need be applied, as the weighting values add to 1. The NWQI is less utilised than numerous other indices, in no small part due to its general nature, which has been noted to cause information loss (due to averaging).

Despite this, there are numerous studies which examine the usage of this index in various scenarios (Wills & Irvine, 1996). The index is a general index of water quality rather than a specialised index, but it is still indicative of water quality behaviour. Further, it is relatively simple to calculate when in possession of the correct sub-index curves and weighting values, with online and Excel™ based calculators increasing the ease further. As such, it can be said that to some extent, the problems commonly experienced with the NWQI are mainly due to its generality; however, this generality and ease of use is what gives this index its greatest advantages. This does not by any means indicate that the index is perfect, nor does it indicate that it will behave in line with expectations.

CWQI

The CWQI was developed by the Canadian Council of Ministers for the Environment in an attempt to create a unified national index to replace the jurisdictional indices in use across certain provinces of Canada (Neary et al., 2001). The equation is defined as follows:

$$CWQI = 100 - \frac{\sqrt{F_1^{\,2} + F_2^{\,2} + F_3^{\,2}}}{1.732}$$

Where:

F_1 = representing scope, the proportion of parameters that exceeded the guidelines;

F_2 = representing frequency, the proportion of failed tests; and

F_3 = representing magnitude, the level by which failed tests exceeded the guidelines.

Where:

$$F_1 = \frac{Failed\ parameters}{Total\ parameters}.100$$

$$F_2 = \frac{Failed\,tests}{Total\,tests}.100$$

$$F_2 = \frac{nse}{0.01nse + 0.01}.100$$

In calculating F_3, *nse* represents the normalised sum of excursions, and it is calculated first by measuring the extent of each excursion using the following equation (a reversed fraction is used in the event of a minimum value):

$$Excursion = \frac{Failed\ test\ value}{Guideline\,value} - 1$$

Then normalised sum of excursion was calculate using the equation:

$$nse = \frac{\sum Excursions}{Nil\ Tests}$$

All three of these variables are calculated individually into a 0-100 rating, with higher levels representing poorer water quality outcomes. Compared to the NWQI, the CWQI has found much more widespread usage, most likely due to its adaptability. Whilet the past studies were able to apply the CWQI with varying degrees of success, a great many of them also noted one of the most reported problems with the CWQI; the undue weight given to the F1 variable.

Effectiveness of water quality indices

Based on the current literature, it becomes apparent that the CWQI is more widely used than the NWQI, due in no small part to its flexibility around both parameters and intended usage, which allows it to adapt to the needs of the study at hand. However, this flexibility is also a factor that can affect the veracity of any one study's results, as it leaves the index extremely open to manipulation. Whilet in the right hands this manipulation can be a tool to better address a study's objectives, or prevent local water quality characteristics (such as higher sediment loads) from adversely impacting upon results, it also opens the index up to a considerable level of bias. Additionally, Whilet this index is relatively accessible, it takes a long time to calculate, most notably due to the multi-step calculations used to create the F3 variable. Finally, as with any water quality index, there is an inherent loss of information during the calculation of this index.

Thus, neither the NWQI nor CWQI are perfect tools for assessing water quality. As such, it becomes necessary to compare their relative accuracy, and their efficacy in expressing water quality data, to determine how best to apply them. A key part of this process is to examine current literature comparing the two index models. However, studies comparing different index models are rare, in part due to the drive to create more accurate indices, as in the works of Kannel et al. (2007), who developed an index based on dissolved oxygen, Bonanno & Giudice (2010), who attempted to create a Floristic Water Quality Index (an index based on local flora), and Nikoo et al. (2011), who attempted to create a probabilistic water quality index using the outputs of the NWQI and CWQI. Additionally, many studies that do compare indices do so to justify either methodology, or the creation of a new index. Tyagi et al. (2013) reported advantages and disadvantages of multiple indices through a comprehensive review of the literature surrounding them. It was found that no index was universally accepted as accurate and that all indices investigated had flaws equal to their strengths.

The main strengths of the NWQI include its ability to summarise data in a rapid and reproducible manner, and the ease with which the concept could be explained to the layperson. However, it was also determined that the index was too general for water quality monitoring, and was unable to adapt to complex environmental issues. In addition to that, it was also noted that there was data loss during handling. According to Lumb et al. (2012), the NWQI provided poor accuracy in its water quality classification, assigning incorrect 'good' or 'excellent' scores. As an example, it was found that the NWQI presented scores of good and excellent in water bodies impacted by anthropogenic activities, including those that ran through high productivity agricultural areas and townships; areas which, according to current literature, should possess relatively degraded waterways (Paul & Meyer, 2001, Walsh et al., 2005, Tippler et al., 2012). The most likely cause of these high results was determined to be that the method used by the NWQI to aggregate the values of the individual parameters followed linear aggregation. In contrast, the CWQI was found to be much stricter, and in almost all cases produced a lower index score. Lumb et al. (2012) attributed this to the non-linear aggregation that the CWQI used.

Finally, there was the study performed by Dede et al. (2013) to compare various water quality indices in relation to a real-world situation. Whilet this study did not utilise the complete NWQI, one of the models it used, dubbed the Universal Water Quality Index (UWQI) was calculated in the same manner as the NWQI, with the only changes being to the sub-index changes. However, even with literature pointing to the CWQI as the better model, this does not mean it is the better model in all circumstances. For example, the index can become unstable with fewer parameters (CCME, 2006) and the F_1 variable is known to affect accuracy and dominate the index. Additionally, fewer factors may improve the accuracy of the NWQI, by lessening the eclipsing of each factor. Therefore, it is unknown which of these indices is preferable in such a situation and as such it becomes necessary to test them in a real-world scenario.

STUDY AREA

Data for selected water quality parameters was collected at five sites based around tributaries in the Upper Parramatta River in Western Sydney, Australia (Figure 1). At each site, physiochemical data was collected above, within, and below either the tributary or its confluence. Bacteriological and macro-invertebrate data was collected only at those points above and below confluence. The results of this data collection were used to assess water quality and the effects of Urban stream syndrome at each site. The five tributaries used in this study were:

- Quarry Branch Creek: Quarry Branch Creek was the furthest upstream site and acted as close to a neutral sample as was possible.
- Finlaysons Creek: Finlaysons Creek was the most developed of the tributaries used in this study.
- Hunts Creek: Hunts Creek was included in this study to provide an upstream tributary to Darling Mills Creek.
- Darling Mills Creek: Darling Mills Creek connects with Toongabbie Creek to form the Parramatta River.
- Domain Creek: Domain Creek runs through Parramatta Park, terminating in a series of weirs.

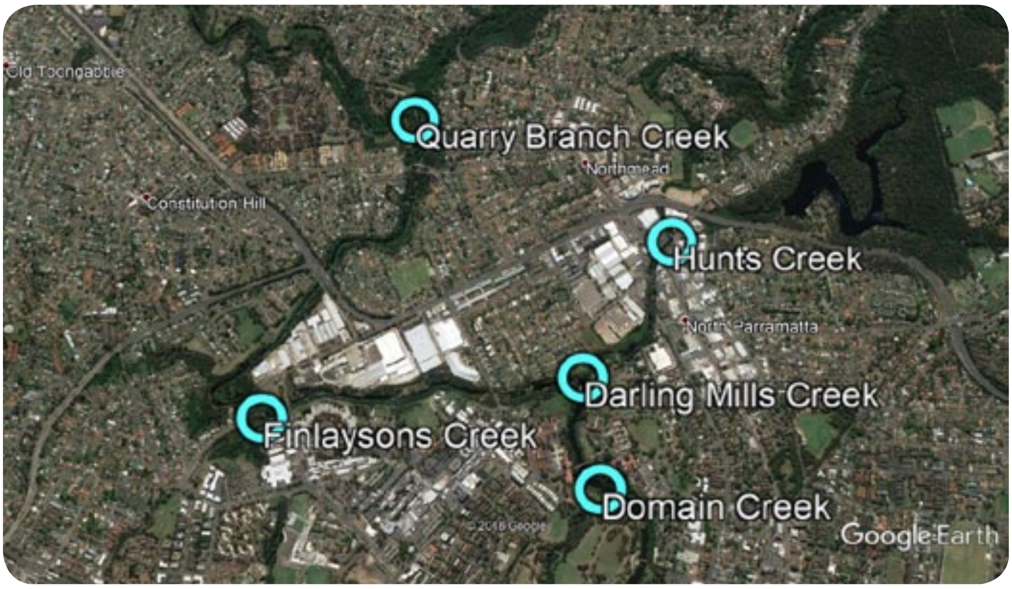


Figure 1. Map of Sample Areas. The above map showcases the study areas used in this study. Study areas match up with the points used in other figures.

Physiochemical sampling was performed on these sites six times throughout the data collection period with one sampling date in May, three in July, and two in August. Bacteriological testing for faecal coliforms, total coliforms and *E. coli* was performed once in July as a group and once on separate dates in August. Macro-Invertebrate surveys were performed prior to the beginning of data collection and following the end of data collection. The parameters used in data collection are categorised as either. physiochemical and biological parameters and were selected both for their ubiquity and their utility as water quality indicators.

Physiochemical Parameters

Seven physiochemical parameters were selected for use in this study. The parameters were chosen for their capacity as an indicator or their inclusion within either the NWQI or the ANZECC guidelines used to calculate the CWQI. Dissolved oxygen, electrical conductivity, total dissolved solids, turbidity and pH are the key indicators of water quality (Sanchez et al. 2007; Rudolf et al., 2002; Kannel et al., 2012; Tippler et al., 2012 and Wright et al., 2012). Electrical conductivity, total dissolved solids and pH relate to the level of water contamination and catchment imperviousness. Phosphate and nitrates provide a measure of nutrient contamination.

Dissolved Oxygen, pH, total dissolved solids, and electrical conductivity were all taken as in-situ measurements between 10:00 and 14:00. Water samples were also taken at these times, and tested off-site for nutrient content, and turbidity.

Biological Parameters

Biological parameters in relation to this study refer both to the macro-invertebrate surveys performed at the commencement and conclusion of the study and the bacteriological testing performed in conjunction with physiochemical testing throughout the study period. The specifics of each are described below.

Bacteriological populations were tested using methods associated with the reagents colilert-18 (faecal coliforms), colisure (total coliforms, *E. coli*) and enterolert (Enterococci), producing results measured in Most Probable Number (MPN)/100ml. As the IDEXX system has a maximum count for faecal coliforms, instances wherein the population exceeded the scope of the test were given the highest number value that the test could assign, of 2419.6 MPN/100mL. Faecal coliform results were collected on July 2 and August 18, for use in the creation of the NWQI. Total coliform and *E. coli* testing were performed on July 2 and August 5 to provide another non-index indicator of real water quality. Enterococci testing was performed on August 18 both as a bacteriological indicator of water quality and for its relevance to potential recreational usage.

Macro-invertebrate surveys were performed prior to and following the main data collection of this study and used to create SIGNAL scores of macro-invertebrate richness for sites both above and below tributaries. This was done to provide another indicator of water quality against which to test the index, as the use of macro-invertebrates as measures both of water quality and overall river health have been well established (Chessman, 2003, Tippler et al., 2012, Sharma et al., 2006). Macro-invertebrate samples were collected with a 250 μm net, preserved with ethanol, and taken off-site for identification. Results of the survey were used to create a SIGNAL score for each site.

Calculating NWQI

Using Equation (1), the NWQI was calculated for each sample point on each day of data collection. Unlike the full version of the NWQI, only five parameters were used consistently; dissolved oxygen, pH, turbidity, nitrates, and total solids (calculated using TDS in this study). Faecal coliforms were included on the days that they were sampled.

The CWQI was calculated concurrently to the NWQI, with sampling occurring at each sample point on every day of testing. In order to operate in a parameter-limited fashion, six parameters were used in the calculation of this index, below the number recommended by the CCME (2006). Parameters were chosen as indicators and for their inclusion in ANZECC guidelines, and included dissolved oxygen, pH, turbidity, electrical conductivity, phosphates, and nitrates. The CWQI was calculated using Equation (2).

Modified CWQI

In addition to the previously calculated CWQI, a modified CWQI was calculated, with increased limiting values for phosphates and nitrates. This will better reflect the background level determined through data collected at the Quarry Branch Creek site. The new limit values used were: Phosphate: 0.3mg/L and Nitrate 0.6mg/L. This modification is in line with Lumb et al. (2012), who suggested that when using the CWQI in a heavily disturbed waterway one might change limit values based upon the values seen at a relative control point, in this case Quarry Branch Creek data analysis. The analysis that was required by this study falls into two categories; the analysis of sub-catchment imperviousness, and regression analysis of the water quality indices against selected indicators.

Analysis of Upstream Corridor Imperviousness

Sub-catchment imperviousness was determined through geo-analysis using GRASS GIS connected to a MySQL database in three steps; creation of upstream corridors for each study site, overlaying of imperviousness data into created corridors, and querying of newly created overlays for imperviousness data.

First, due to a lack of data representing the sub-catchments of the Parramatta River, this study created corridors for selected tributaries. Corridors were selected for ease of creation and due to the relationship between corridor structure and waterway/ecosystem health (Collier & Clements, 2011). Corridors were created in GRASS-GIS using the v. buffer function, with a distance of 0.065 units, or approximately 650m. This buffer data was calculated to extend completely upstream of each specific data point, forming a value determined to be representative of complete upstream imperviousness. This was done so that favourable local conditions would not obscure the effects of upstream imperviousness. Second, each buffer zone was overlayed upon an imperviousness map provided by Greater Sydney Local Land Services and data imported to create imperviousness maps within the created corridors. Finally, the data from each buffer zone was analysed using MySQL to determine the proportion of impervious surfaces in any upstream corridor, expressed as a percentage.

Regression Analysis

Regression analysis was used to test for significant relationships between the previously generated Water Quality Indices and selected factors, with a 95% confidence limit. The factors selected for regression analysis were SIGNAL score, bacteriological populations, upstream corridor imperviousness and physiochemical parameters. The two SIGNAL scores collected at each site were averaged to give a combined SIGNAL score. Bacteriological population included values for total coliforms, *E. coli*, and enterococci, with regression analysis occurring identically on all three. Upstream corridor imperviousness results were obtained using the above method.

In cases such as physiochemical parameters and corridor imperviousness, regression analysis was performed on all points of data, with constant values such as imperviousness being repeated at the same level on all dates. In regards to SIGNAL scores, regression analysis was only performed for data above and below confluence, in line with where macro-invertebrate surveys were performed. SIGNAL values for sites were given at the correct values on all days of sampling, and presented constantly, similarly to corridor imperviousness. Regression analysis on bacteriological results only used data collected on the day of bacteriological sampling due to the effect that physiochemical parameters can have on bacteriological communities. As this study is working to a 95% confidence limit, a significant relationship is considered to exist when the p-value produced by a regression analysis is less than 0.05.

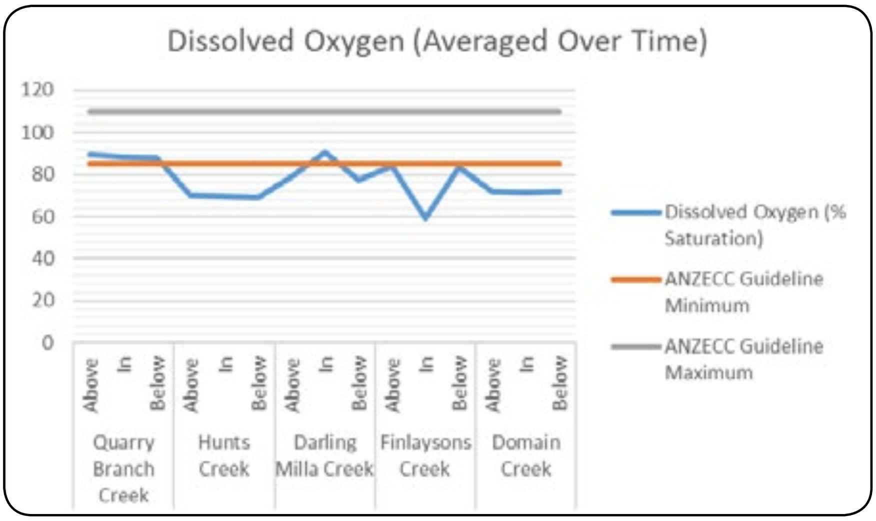


Figure 2. Variation of dissolved oxygen values between sites, averaged over time.
Regression analysis did not show a significant relationship between catchment imperviousness and levels of dissolved oxygen, however such a relationship did exist between dissolved oxygen and date, potentially suggesting that run-off from water quality events or general climactic conditions are driving dissolved oxygen levels.

RESULTS

Dissolved Oxygen

Dissolved oxygen levels were found to be below the recommended guidelines in the majority of tests, with 56 of 90 tests presenting a DO level lower than the 85% saturation recommended by the guidelines (Figure 2). Dissolved oxygen levels for each date of sampling presented as a densely populated range of values presented with multiple outliers of extremely low concentration. An example of this exists in the August 5 data, where Hunts Creek sites present with levels as low as 27% saturation, compared to an average saturation of 63.3% for all samples collected on that date.

Regression analysis against biological indicators showed a significant positive relationship between SIGNAL scores and dissolved oxygen. In relation to bacteriological indicators, dissolved oxygen presented no significant relationship with either Total Coliforms or Enterococci levels, but presented a significant relationship with *E. coli,* with lower levels of dissolved oxygen presenting concurrently with greatly increased levels of *E. coli.*

pH

pH levels were generally found to fall within the ANZECC guideline of 6.5-8, with only 13 of 90 results presenting in excess of either limit (Figure 3). Generally speaking, pH tended towards the basic end of the spectrum, with 50 of 90 tests showing a pH of greater than 7 and the remaining 40 showing a pH lower than 7. This pattern is more severe in relation to excursion events, where nine tests produced pH values higher than the upper limit prescribed by the ANZECC guidelines, and four tests beneath the lower limit. As pH levels generally presented within the guidelines, results mainly occurred within the 6.5-8 range. In addition, excursion events tended to be small in scale, unlike the major outliers that were found in other parameters, suggesting a relatively stable pH throughout the catchment.

Regression analysis showed that pH did not have a significant relationship with catchment imperviousness, however a significant relationship was found with date. In regards to biological indicators, only total coliforms presented a significant relationship with pH, with lower pH levels being associated with higher coliform levels.

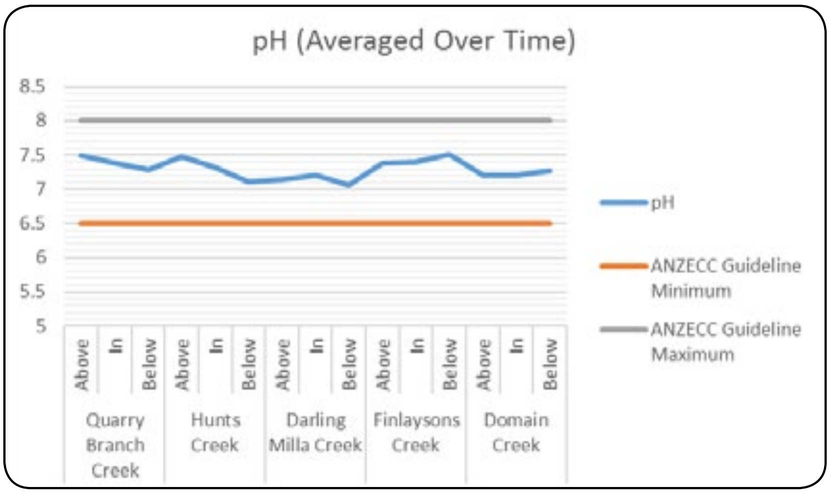


Figure 3. Variation of pH values between sites, averaged over time.

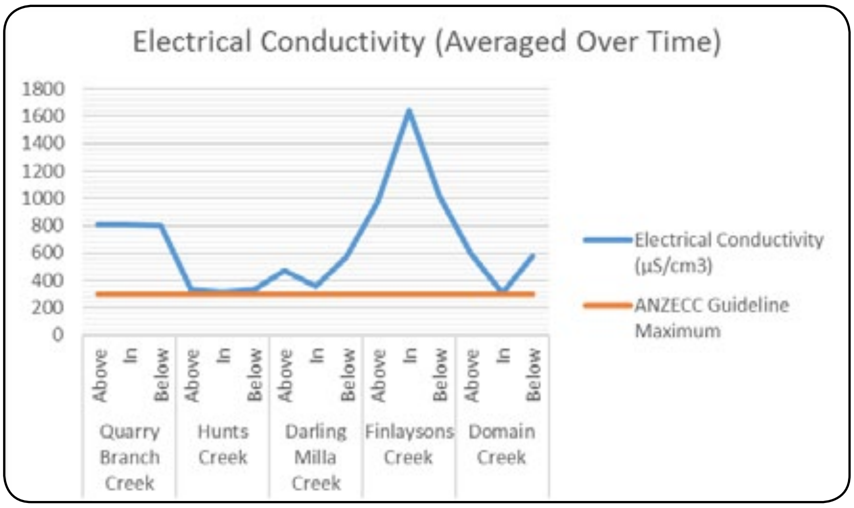


Figure 4. Variation of electrical conductivity values between sites, averaged over time.

One potential explanation of this can be found in the work of Pearson et.al (1987), who found that high pH levels were associated with an increased rate of die-off in coliform bacteria. This was further supported by the work of Curtis et.al (1992), who found that light-induced damage to bacteria was significantly increased in a high-pH environment. However, as both these experiments mainly considered pH ranges above what this study examined, they can be considered to only provide a partial explanation.

Electrical Conductivity and Total Dissolved Solids

As shown Figure 4, electrical conductivity was generally in excess of the guidelines, with 72 of 90 readings taken occurring above the 300 S/cm ANZECC guideline. Electrical conductivity data from each date presented as a general range acting as a background level with several high-conductivity outliers, most notably around Finlaysons Creek which generally posted the highest level of conductivity. Regression analysis showed a significant relationship between electrical conductivity and catchment imperviousness, in keeping with current models of Urban stream syndrome (Paul & Meyer, 2001, Walsh et al., 2005, Tippler et al., 2012). Further regression analysis showed a similarly strong link between electrical conductivity and date of sampling, suggesting that rain events between sampling may have led to a change in electrical conductivity levels, also in keeping with the current knowledge on urban stream syndrome.

In regards to biological indicators, no significant relationship could be found between electrical conductivity and either total coliforms or enterococci, however a significant relationship was found in relation to *E. coli,* with lower electrical conductivity showing a relationship with higher levels of *E. coli* bacteria. A significant relationship also existed between SIGNAL scores and electrical conductivity, with macro-invertebrate communities appearing to favour sites with a higher conductivity level. The behaviour and regression analysis for total dissolved solids mirrored that of electrical conductivity However the strength of relationships was slightly different, with TDS presenting a stronger relationship with *E. coli,* but a weaker relationship with SIGNAL scores compared to electrical conductivity.

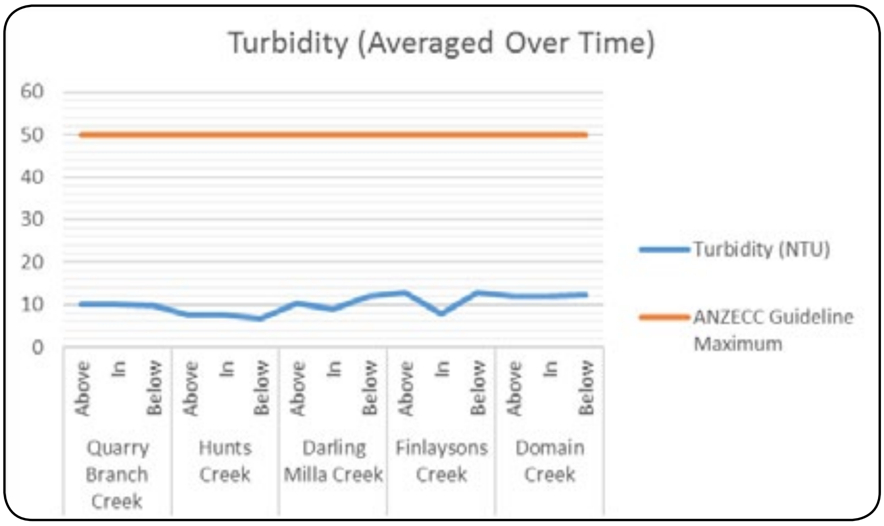


Figure 5. Variation of turbidity values between sites, averaged over time..

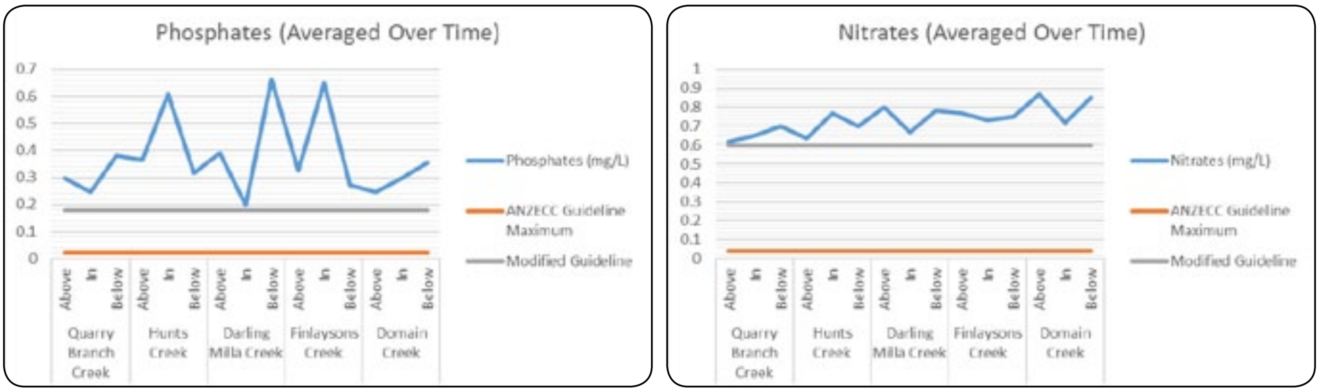


Figure 6. Variation of nutrient parameter values between sites, averaged over time.

Turbidity

Turbidity levels were below the limit of the ANZECC guidelines. Turbidity levels generally presented as a large grouping around the value of 10. Regression analysis showed a significant relationship between turbidity and catchment imperviousness, with higher imperviousness leading to higher turbidity levels. Similarly, a significant relationship could be ascertained between turbidity and date of sampling with a decreasing trend over the study period. A significant relationship also exists between turbidity and SIGNAL score, with SIGNAL score increasing with higher turbidity. This may represent a preference in habitat for slightly higher turbidity levels, a statistical anomaly, or evidence that more natural waterways (with higher turbidity levels) are being favoured by benthic macroinvertebrates.

Neither total coliform nor *E. coli* levels showed a significant relationship with turbidity however such a relationship was found in the case of enterococci. This may be due to the known negative effects of ultraviolet radiation on the vast majority of Faecal Indicator Bacteria (Reed, 1997, Wegelin et al., 1994, Šolić & Krstulović, 1992, Byappanahali et al., 2012), and thus higher turbidity may represent a more suitable environment for growth. Another potential cause is that higher levels of enterococci favour water with a higher concentration of suspended solids, which manifest as turbidity.

Phosphates and Nitrates

Levels of filterable reactive phosphate were universally above the level recommended by the ANZECC guidelines, with a minimum level of 0.18 mg/L compared to the 0.02 mg/L level ascribed to lowland rivers in the guidelines (Figure 6). Phosphate levels generally presented at a fairly uniform level across sites, with the exceptions of high-level outliers. Regression analysis performed on phosphate levels showed no significant relationship with catchment imperviousness. However, it did present a significant relationship with date, once again suggesting that rain and general weather events are a major determining factor.

In relation to biological indicators, neither SIGNAL scores nor bacteriological indicators displayed any significant relationship with phosphate levels. Similarly to filterable reactive phosphate, nitrate levels were universally in excess of the ANZECC guidelines, with a minimum score of 0.2mg/L to the recommended level of 0.04 mg/L, and generally presented as clusters accompanied by outliers.

Regression analysis showed that nitrates did not possess a significant relationship with catchment imperviousness but a significant relationship was found in regards to date, much like in the case of phosphates. Nitrate levels were not found to have a significant relationship with either SIGNAL scores, total coliforms, or *E. coli,* but they were found to have a significant relationship with levels of enterococci bacteria. This suggest that enterococci rather than other bacteriological indicators have a preference for high-nutrient environments.

Biological Indicator SIGNAL Scores

SIGNAL scores were calculated for each site based on all collected data, with commencement and conclusion surveys calculated as a single score. These scores were then assigned to their relevant site on all sampling days. SIGNAL scores were generally low, with a minimum result of around 3.2 below the Hunts Creek confluence and a maximum result of 4.67 above Quarry Branch Creek. The commencement survey found small macro-invertebrate populations across most sites, with the exception of Finlaysons Creek and Hunts Creek, where high numbers of *Gastropoda Lymnaeidae* were found above confluence. With the exception of *Gastropoda Lymnaeidae,* no one species was found to dominate results.

The macro-invertebrate survey found relatively large macro-invertebrate populations at most sites, however diversity within this population was relatively low, with a limited number of families found. *Cladocera Moinidae* was found across all sites, with the highest population being found in the Domain Creek sites. The other dominant family was *Plecoptera Notonemuridae*, which was found across the majority of sites, and in high numbers above both Darling Mills Creek and Finlaysons Creek. Regression analysis showed a significant positive relationship between catchment imperviousness and SIGNAL scores, potentially due to the link between catchment imperviousness and electrical conductivity; another apparent positive factor in relation to SIGNAL scores. As has already been discussed, electrical conductivity, total dissolved solids, turbidity, and dissolved oxygen were all found to have a significant and positive relationship with a sites SIGNAL score, the potential causes of which have already been raised.

Faecal Coliforms

Faecal coliform levels on the first day of bacteriological sampling were extremely high; most likely due to rainfall one day before sampling. However, the number of results exceeding the limits of the Colilert testing method on the July 2 sample date rendered that data unfit for statistical analysis. As such, regression analysis was only performed on faecal coliform results from August 18. In relation to the ANZECC Guidelines, levels of faecal coliform bacteria universally exceeded levels considered acceptable for primary contact. In regards to secondary contact, both Quarry Branch Creek and Domain Creek results presented at acceptable levels with all other sites presenting at a rate greater than 1000 MPN/100mL.

No significant relationship was able to be found between faecal coliform levels and catchment imperviousness. However, a significant relationship was found with dissolved oxygen, pH, total dissolved solids, and electrical conductivity. With the exception of pH, these relationships were found to be universally negative. In the case of dissolved oxygen, the work of Reed (1997) provides an explanation, as dissolved oxygen was found to increase the effects of solar radiation, and shown by Šolić and Krstulović (1992) to have a negative effect on faecal coliforms.

Total Coliforms

As total coliform levels are not used in ANZECC water quality guidelines, no comparisons were made against guidelines. Total coliform levels averaged at a level of around 1400 MPN/100 mL, with slightly higher levels found on August 5 than July 2. Regression analysis showed no significant relationship to catchment imperviousness, however a relationship was found against pH, with slightly lower pH translating into higher total coliform counts. As has already been noted, the reason for this may be a preference for low pH within coliform bacteria, to which faecal coliforms represent a notable exception, *E. coli.*

Whilet *E. coli* is generally noted as being a good indicator of water quality for recreational use within freshwater ecosystems not enough research has been done to properly form a guideline value. As such, *E. coli*, like faecal coliforms, was used as an indicator rather than a clear limiting value. *E. coli* levels were found to have a median of around 190 MPN/100mL. *E. coli* populations tended to be lower on average on August 5 than July 2; however, August 5 also produced the highest measurements of *E. coli*, with both Hunts Creek sites presenting at levels around 1000 MPN/100mL. Regression analysis showed a negative relationship between catchment imperviousness and *E. coli* levels, with higher catchment imperviousness generally decreasing population levels.

In regards to physiochemical parameters, *E. coli* showed significant relationships with dissolved oxygen, electrical conductivity, and total dissolved solids, with these relationships being universally negative. This is especially noteworthy in the case of dissolved oxygen, as it could represent a causative link, either due to microbiological levels depleting dissolved oxygen levels, or of *E. coli* thriving in an environment lower in dissolved oxygen. The significant relationship with dissolved oxygen can be partially explained by the findings of Reed (1997), where dissolved oxygen was found to augment ultraviolet toxicity within both *Escherichia Coli* and *Enterococcus Faecalis.* In relation to electrical conductivity, the works of Carlucci & Pramer (1961) showed a negative relationship between bacteria survival and salinity levels (analogous of electrical conductivity).

Enterococci levels generally exceeded the guidelines for primary recreational contact, with the exceptions of the Domain Creek sites and below confluence at Darling Mills Creek. However, with the exception of the site above the Hunts Creek confluence, levels universally fell within ANZECC guidelines for secondary recreational contact. Generally speaking, enterococci levels were highly varied, with no major grouping occurring. Regression analysis showed no significant relationship between Enterococci levels and catchment imperviousness. A negative relationship was found between nitrate and enterococci levels, and a positive relationship was found between turbidity and levels of enterococci, likely as a result of ultraviolet toxicity (Byappanahalli et al., 2012).

WATER QUALITY INDICES

CWQI

The CWQI universally produced results in the "Poor' and 'Marginal' range, with 74 'Poor' (<44) and 16 'Marginal' (>44, <59) results. Results mainly presented between 20 and 40, with the highest concentration of results being in the 30s (Figure 7). Five outliers existed below 20, and four above 50. Regression analysis between index scores and catchment imperviousness revealed a significant negative relationship, as was expected. Additionally, regression analysis between the index and its component parameters showed significant relationships with four of the six parameters used.

The first of these was dissolved oxygen, which as expected showed a positive correlation with index scores. The importance of this relationship can be linked to the already established properties of dissolved oxygen as a general measure of water quality. The second physiochemical parameter that presented a significant relationship with the CWQI was pH, with the relationship being negative. This result can be viewed in two ways. First, it could be argued that this points to the river's natural pH being slightly acidic and that an unknown contaminant or factor is increasing the pH unnaturally. The second way this result can be viewed is that environmental factors that increase the pH of the river also increase the values of other parameters. This viewpoint in solidified when the reaction of pH levels to date is considered.

The index also presented significant relationships with phosphates and nitrates, both of which were negative in nature, suggesting that higher nutrient loads are either symptomatic of, or a direct cause of poor water quality within the river. In regards to biological indicators, significant relationships were found with total coliforms, *E. coli* and enterococci. However, these relationships were universally positive, suggesting either that insufficient bacteriological data was collected for accurate analysis, or that bacteriological indicators have a strong negative correlation to markers of poor water quality, such as pH and electrical conductivity.

Modified CWQI

Once the CWQI was modified to be more representative of background nutrient levels, the scores of the index were much more evenly distributed over the scale and appeared to be much more representative of the health of the river. The main score distribution settled between scores of 50 and 70; there were outliers with one score in the 20-30 range and four scores in the 80-100 range. Regression analysis with the modified index showed a strong significant relationship against catchment imperviousness. The relationship was found to be negative, in line with both expectations and current knowledge of water quality behaviour.

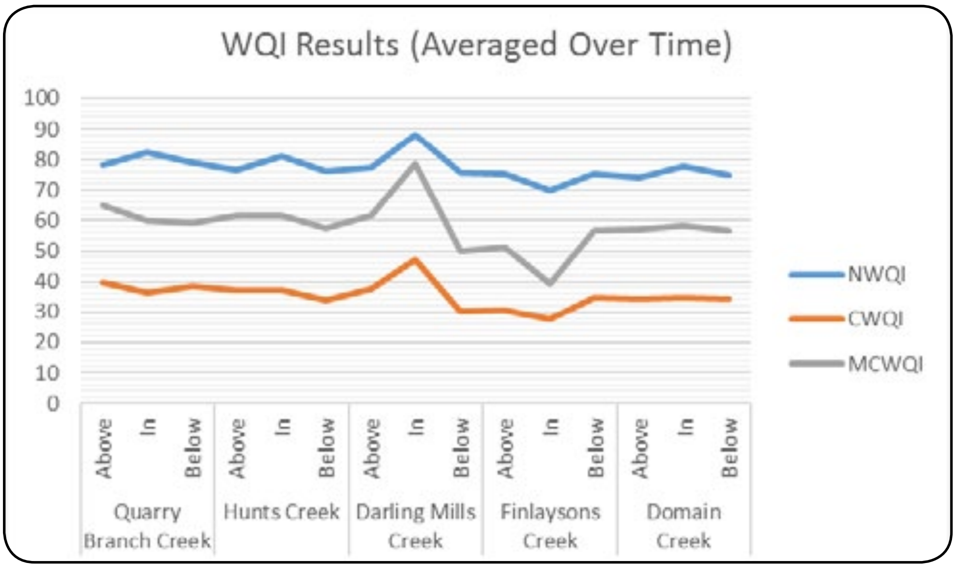


Figure 7. Variation of NWQI, CWQI and modified CWQI values between sites, averaged over time.

The modified CWQI was also found to possess significant relationships with all physiochemical parameters except turbidity. Relationships were negative for all parameters except dissolved oxygen, matching both expectations and the behaviour of the CWQI. Similar to the CWQI, the modified CWQI found significant relationships with total coliforms, *E. coli,* and enterococci; however, these relationships were found to be positive.

NWQI

The NWQI presented higher results than either formulation of the CWQI, with the majority of results above 80. Not only are these results higher than the other indices, they are also not representative of the actual state of water quality in the study area. Regression analysis performed on the index showed a significant relationship with both catchment imperviousness and date, suggesting a relationship with both catchment imperviousness and environmental factors such as rainfall.

In regards to physiochemical parameters, regression analysis showed significant relationships with dissolved oxygen, electrical conductivity, and total dissolved solids. Additionally, these relationships were found to be stronger than their counterparts in the CWQI. In part, this can be attributed to the inclusion of dissolved oxygen (DO) and total dissolved solids in the indices, calculation, with electrical conductivity showing a relationship as a result of its significant relationship with total dissolved solids (TDS).

Biological parameters generally tested negative to a significant relationship with the Water Quality Index, with the exception of *E. coli*, which had a negative relationship with the index (increased levels of *E. coli* heralding a lower index score) and is most likely due to its relationship with TDS and DO. This may suggest either that the index is inaccurate in regards to the expected behaviour of biological indicators or that *E. coli* is the most consistent biological indicator in regards to water quality parameters. Faecal coliform levels were not tested for significance, as they were used in the calculation of the index.

DISCUSSION

Comparative Accuracy of Indices

In order to compare the accuracy of the three indices it was first necessary to determine a method by which accuracy within the study could be measured. The eventual method settled upon was to determine both the overall accuracy of the indices used and test their responsiveness to known indicators and causes of poor water quality. In order to do this, the indices were evaluated in regards to three factors.

The first factor considered in regards to accuracy of these indices was the level to which they accurately represented the overall water quality of the study area. In order to evaluate this, index values were converted into a 1-10 matrix and measured to determine which indices had the greatest overlap with SIGNAL scores, as opposed to a fully formed relationship. This was accomplished by utilising the highest and lowest SIGNAL scores across all sites to create a 'range' within which index results would be expected to fall. The NWQI performed the worst in regards to this test, with all index scores found to be above the range indicated by SIGNAL scores. This severely limits the utility of this index as a tool to convey study results, especially in cases where parameters are limited.

The CWQI was much more accurate than the NWQI, with 23 of the 60 results within the range suggested by SIGNAL scores. However, those results not falling within the expected range were below the range, showcasing the low scoring nature of this index when used in a limited capacity. This lines up with the findings of Lumb et al. (2012), who found that the CWQI consistently presented lower scores than US models of water quality indexing. In part, this may be due to the tendency of the CWQI to give too much weight to its F_1 or 'scope' factor, which measures the proportion of failed parameters within the selected set (CCME, 2006). Within a study such as this which uses a limited selection of parameters, this inaccuracy has the potential to be even greater, as the lower number of parameters increase the effect of each failed parameter on the site's relative F_1 value. Of all the indices studied, the modified CWQI performed the best in regards to expected water quality, with 39 of the 60 index scores used in this evaluation lying within the expected range. In addition to this, those results lying outside of this range were evenly split, with 11 presenting scores higher than the expected range and 10 presenting lower. With this in mind, the modified CWQI is the more accurate index for public information about environmental water quality.

The second factor used in determining index accuracy was the behaviour of the indices in regards to a known factor affecting water quality; catchment imperviousness. To determine accuracy in regards to this factor, the results of the relevant regression analysis were examined and a rank given to each index, dependent first on the existence of a significant relationship (p-value < 0.05), and then on the strength of their relationship, determined by which factor had the lowest p-value.

The CWQI had the weakest relationship with catchment imperviousness at 0.038. However, this result was only slightly lower than its next competitor, the NWQI, which possessed a p-value of 0.036. The fact that both perform comparably is not altogether surprising, as electrical conductivity and TDS both strongly correlate with catchment imperviousness, as well as comprising a large part of both respective indices. The modified CWQI showed the greatest relationship with catchment imperviousness, presenting with a p-value of 0.002. Based on this metric, the modified CWQI is the most accurate index in regards to current models of water quality behaviour and as such can be considered an accurate tool to measure water quality trends within a limited water quality investigation.

Finally, indices were tested for significant relationships to selected bacteriological indicators, in an attempt to capitalise on the usefulness of certain bacteria in indicating both poor water quality and the faecal contamination of water (Byappanahalli et al., 2012). In regards to this test of accuracy, the NWQI performed the worst, correlating only with faecal coliforms (which were integral to its creation), and *E. coli.* Both of these relationships were found to be negative in nature, with higher levels of bacteria leading to lower water quality index values.

The CWQI and modified CWQI correlated with three out of four bacteriological indicators; total coliforms, *E. coli*, and enterococci. However, the nature of these relationships was positive, with higher bacteriological populations unexpectedly correlating to higher water quality index scores. The most likely causes of this discrepancy are a lack of sampling data, high WQI results on sampling days, and a negative relationship between certain physiochemical parameters with bacteriological indicators and water quality index values.

Based on these results, it can be broadly stated that neither water quality index model performed well in regards to bacterial populations, a fact that raises major questions about both their accuracy and their utility in scientific study. These questions can be considered in a number of ways.

First, one can consider its failure to show a significant relationship with bacteria indicating it lacks accuracy, as many of the bacteria tested for are considered Faecal Indicator Bacteria (US EPA, 2012). Based on this result none of the indices provide enough accuracy, suggesting that a water quality index is not a suitable tool to use in a study with limited parameters. However, this goes against one of the claims related to the CWQI; that it can present a relatively accurate result with only four parameters, so long as four or more monitoring events are calculated (Neary et al., 2001).

The second way that the lack of accuracy regarding bacteriological factors can be considered is a failure in bacteriological testing, with the study requiring a larger dataset to determine any significant link between bacterial populations and water quality. If this is the case, then it suggests that bacteriological indicators are unsuitable for use in a limited study unless the express purpose of the study is linked to said populations as in the case of recreational water quality.

The final way that one can consider the failure of the indices is the result of their misapplication, with the indices better suited to provide insight into the environmental and physiochemical factors that may be affecting the river, rather than bacteriological or health-based factors. This viewpoint relates to the application and suitability of the Water Quality Index.

Suitability for Limited Water Quality Monitoring

Based on the results found in this study, the usefulness of any of the created indices is extremely dependent on their intended application, as is whether they are applicable in relation to a limited water quality investigation.

NWQI

In relation to this study, two flaws became apparent in regards to the NWQI. Firstly the quality curves for certain parameters were not in line with ANZECC levels for environmental protection and secondly, the expected results are not of high quality. An example of this is in DO, where the limit value imposed by the ANZECC guidelines (85% saturation) presents a quality index of 91. Another example of this is nitrates. Within the ANZECC guidelines, a limit of 0.04 mg/L is placed on nitrate levels; however, the NWQI gives 1 mg/L nitrate levels a quality index of 96. The effects of this are compounded when the second major flaw of the NWQI is considered: loss of information.

Loss of information is a commonly held issue with the NWQI (Tyagi et al., 2013, Wills & Irvine, 2011) and represents the ability of a few extremely high or low parameter scores to create false results. An example given by Wills & Irvine (1996) found that if all other parameter measures were pristine, a pH sub-index score of zero would only lower the result to 85, still within the 'good' class of water quality despite being unsuitable for certain forms of aquatic life, drinking and recreation.

When both of these flaws are viewed together, it provides a potential explanation for the abnormally high results many sites were experiencing, as high Qi's for various parameters led to the overall index scores being higher than expected.

Due to the flaws in the NWQI, it has little use as a tool to measure water quality, as data loss and optimistic Q_i scores negatively affect the accuracy of the index, with limited studies suffering more due to the lower number of parameters. In addition to this, Whilet the NWQI showed a significant relationship to catchment imperviousness, suggesting utility as a tool to evaluate the effect of environmental factors, this relationship did not survive when NWQI values were averaged out over time, suggesting an underlying inaccuracy. In order to make the NWQI more accurate, the curves that it uses should be updated, with new sub-indices created in response to specific situations and intended uses of the index. In addition, studies should, where possible and appropriate include testing of Biological Oxygen Demand (BOD) if they plan on using this index in any capacity.

CWQI

One of the greatest issues reported with the CWQI is that the influence of the scope value F_1 is too great (CCME, 2006). Additionally, this effect is found to increase in cases with low numbers of parameters, such as in our study, or where poorly selected parameters are used (Terrado et al., 2010, CCME, 2006). In order to determine whether this was a factor limiting the accuracy of the CWQI within this study, correlation analysis was performed on both the CWQI and modified CWQI against its component features. The results of this analysis (Table 1) were normalised on a 0-1 scale, with 1 representing a complete correlation, as would exist between two data-sets of complete covariance, and 0 representing a complete lack of correlation. In cases of a negative relationship, the scale is represented approaching -1.

Table 1. Correlation of variables CWQI.

	F1	F2	F3	0.0000
F1	1.0000			
F2	0.9715	1.0000		
F3	0.1149	0.0843	1.0000	
CCME WQI	-0.9403	-0.9350	-0.4110	1.0000

Results of correlation analysis for the CWQI found that both F_1 and F_2 scores were heavily correlated to the value of the index, Whilet correlation with the F_3 value for magnitude was relatively low. This not only presents a probable cause for this index's poor accuracy within this study, but it also backs up what was found by Lumb et al. (2012), Terrado et al. (2010), and the CCME (2006), in regards to F_1 having too great a role in the final calculation of the Index.

The modified CWQI presented similarly (Table 2) to the CWQI in terms of correlating with variables, with two noteworthy variations. First, the extent to which the F_1 and F_2 variables correlated to the index was lower, and the correlation of the F_3 index higher, suggesting a more even correlation between the variables than in the CWQI. Second, the F_2 index presented as the dominant variable, rather than the F_1 suggesting that the effect of the F_1 variable is somewhat mitigated when using locally formulated guidelines.

Table 2. Correlation of variables for modified CWQI.

	F1	F2	F3	MCCME WQI
F1	1.0000			
F2	0.8845	1.0000		
F3	0.1367	0.202344747	1.0000	
MCCME WQI	-0.9082	-0.9391	-0.4525	1.0000

Another issue reported in regards to the CWQI is its behaviour when parameters are low in number or poorly selected. As such, its suitability in studies using limited water quality data may be low. As was reported in the CCME sensitivity analysis of the index, Painter & Waltho (2003) determined that the CWQI performed in a reasonably stable and accurate way with the use of 10 parameters and that further parameters increased accuracy at 21 parameters and 32 parameters respectively. However, the CCME report refutes this, noting that the parameters selected were of greater importance than the number of parameters, and that the index performed reasonably when computed with as few as seven parameters, the minimum number it recommends. However, this study used a six-parameter index, which Whilet above the four-parameter minimum put forth by Neary et al. (2001), still represents a lower-parameter calculation of the index than is recommended for accuracy.

It can be argued that the necessary number of included parameters is dependent upon the intended use of the index. As an example, a study such as that of Rickwood and Carr (2009) or Khan et al. (2004), which uses the CWQI to measure the suitability of drinking water will be more focused on specific contaminants, and as such will require a larger number of parameters to produce an accurate result. A study more focused on environmental water quality, however, will be more interested in parameters such as DO, which act as overall indicators of water quality, and therefore may be able to operate accurately with fewer parameters. Within the context of this study, both forms of CWQI proved relatively accurate for day-by-day modelling with limited parameters; however, the modified index was more accurate.

Overall, this study recommends that more research is required in determining how best to undertake Water Quality Index creation, how many parameters are required to create an accurate result, and how to best determine the parameters that should be included for a specific site. If a viable limited-parameter index can be created, then it could be used to streamline Water Quality Index creation, and allow water quality studies with few resources to compile their results in a relevant format. Additionally, this would make the process more accessible to citizen scientists, potentially providing a greater data-set on which to base future studies.

This study also recommends that work be undertaken to develop an Australian Water Quality Index based around the CWQI, as a tool for public education and the identification of trends. Any index created should have the goal of providing accurate results with a single day of sampling, which would allow results to respond to factors such as run-off, and allow its use in conjunction with automated water quality sampling, as suggested by Terrado et al. (2010). The development of such an index would provide a useful public information tool, as well as faster analysis of scientific results and improve communication about water quality issues in Australia.

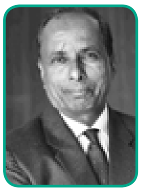
CONCLUDING REMARKS

This study compared the accuracy of three water quality indices created from data collected in the Upper Parramatta River, by comparing them to the sensitivity of local macro-invertebrate communities, catchment imperviousness and selected bacteriological parameters. The indices tested were a parameter-limited NWQI and CWQI, the latter of which was used to produce a modified CWQI.

It was found that the CWQI was more accurate index in terms of both general representation and behaviour in relationship to specific parameters, but only when calculated with parameter limit values modified, as these values were in excess of current guidelines. Without this modification, the CWQI was found to produce lower than expected index scores, and show no relationship to catchment imperviousness. Whilet the NWQI generally behaved within current models of water quality, it scored much higher than either the CWQI or its expected range. The NWQI also showed a weaker relationship to catchment imperviousness when compared with the modified CWQI. Neither of the indices showed widespread correlation against bacteriological parameters, with the CWQI and NWQI only showing a relationship to one parameter each. This suggests that the level of bacteriological testing performed in this study was insufficient to determine its relationships with other factors. As such, further study is recommended in this area.

Based on the results of this study, neither the CWQI nor the NWQI can be suggested as a single measurement of water quality, especially within the context of the current water quality study. However, this study does endorse the use of a locally-modified CWQI as a tool for public education and the determination of general water quality patterns due to its accuracy in both general water quality determination and response to environmental variables. Additionally, should an Australian water quality index be developed for purposes of pattern illustration and public recognition, it is suggested that the framework be based on the CWQI which provided results more in line with expected water quality and, when appropriately modified, fits well with current models of water quality behaviour.

THE AUTHORS



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Professor Maheshwari has over 30 years' professional experience with a particular research focus on peri-urban water management, groundwater and surface water management, water recycling, river health, landscape irrigation, regional water resources planning and sustainability.

During the last ten years, he worked on trans-disciplinary and participatory approaches to water research and focused on understanding how both surface and groundwater, landscape and people interact and influence the environment, sustainability and the local and regional water security. His work involved modelling and analysing the water cycle for long-term water resource planning at the village, metropolitan and regional level and examining the implications of social, economic, cultural, policy and institutional aspects of water cycle management. An important learning out of his research during the last 15 years is that engaging community, government agencies and other stakeholders is critical to address significant, local water and sustainability challenges. He has over 200 publications, including over 90 articles in peer reviewed international journals.

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