The Effectiveness of Global Constructed Shallow Waterbody Design Guidelines to Limit Harmful Algal Blooms

S. Liu1, F. Johnson1, B. Tamburic1, N. D. Crosbie1,3, and W. Glamore4

1Water Research Centre, University of New South Wales, Sydney, NSW, Australia, 2Applied Research, Melbourne Water Corporation, Docklands, VIC, Australia, 3Faculty of Engineering, University of New South Wales, Sydney, NSW, Australia, 4Water Research Laboratory, University of New South Wales, Sydney, NSW, Australia

Abstract

Constructed shallow waterbodies are often designed and built to limit harmful algal blooms in urban regions. Efforts to reduce algal bloom occurrence in these waterbodies have largely focused on waterbody design, catchment criteria and onsite engineering options. However, many constructed shallow waterbodies that comply with design guidelines still experience harmful algal blooms. Identifying the knowledge gaps in current guidelines and examining their recommended design criteria can improve their effectiveness to reduce algal outbreaks. Here, we reviewed 66 global guidelines and identified common design criteria. The use of a ‘one size fits all’ empirical approach and dated literature are common issues associated with the design criteria recommended. Further, only approximately one third of the guidelines that were analyzed directly mentioned harmful algal bloom-related design criteria. To test the validity of these design values in a real-world setting, the suitability of design factors in limiting harmful algal blooms was assessed by analyzing 222 shallow waterbodies monitored over a 9 year period in southeastern Australia. The site analysis indicated that macrophyte area to surface area ratio, shoreline development index, and fetch are the three most influential single design factors associated with harmful algal bloom reduction. The analyses highlighted the ineffectiveness of the existing design criteria globally, with blooms occurring even though some waterbodies were designed in accordance with recommended parameters. The analysis suggested that understanding interactions between multiple design factors may be a useful approach, for example, when considering the macrophyte area to surface area ratio in combination with the shoreline development index.

Plain Language Summary

Constructed shallow waterbodies, including wetlands, lakes and ponds used for stormwater management, are vulnerable to harmful algal blooms (HABs). Appropriate engineering design is important to reduce the risk of HAB occurrence in these waterbodies. A review of current design criteria in these guidelines was conducted to understand the recommended controls for HABs and whether sufficient design advice is available. The effectiveness of these criteria was then tested by statistically analyzing over 200 waterbodies in southeastern Australia over 9 years. Comparing the guideline review and the data analysis indicated that three lake factors, including macrophyte area to surface area ratio, shoreline development index and fetch, could reduce HAB risk. It also showed that the current design criteria may not be adequate to minimize HABs. This study provides a practical approach for investigating waterbody design guidelines and is important for water managers, government staff and industry consultants involved in planning activities related to harmful algal blooms.

1. Introduction

Cyanobacterial blooms, or harmful algal blooms (HABs), have been an increasing threat for the management of freshwaters worldwide (Carmichael, 2008; Smaida, 1997; Watson et al., 2015). Excessive cyanobacterial growth in these waterbodies has been associated with increasing population growth, nutrient sources dominated by urban and agricultural runoff (Bullerjahn et al., 2016) and climatic shifts (Havens et al., 2016). HABs can have adverse impacts on the environment and can be a risk to human health (Bláha et al., 2009; Saqrane & Oudra, 2009; US EPA, 2013). Cyanobacterial blooms are also commonly found in constructed waterbodies (Brasil et al., 2016; de Castro Medeiros et al., 2015; Jayatissa et al., 2006; Shaw et al., 1999).
Constructed waterbodies, including wetlands, ponds and lakes, which receive stormwater and local runoff are increasingly constructed in residential and urban areas (Watson et al., 2015). Constructed waterbodies are often built for flood attenuation, water quality enhancement (Kadlec & Wallace, 2008), providing habitat, water recycling potential, and improving landscape amenity (Melbourne Water, 2005b, 2005c; Sharma et al., 2018). Regardless of the design purpose, eutrophication that results in HABs, is one of the least desirable consequences for the local community (L’Ecuyer-Sauvageau et al., 2019; Watson et al., 2015). Therefore, appropriate engineering design of these waterbodies is critical to reduce the likelihood of HABs (Gibbs & Hickey, 2012; Melbourne Water, 2005a; Mitsch, 2017; Wahlroos et al., 2015).

Internationally, various guidelines and design criteria have been developed for shallow constructed waterbodies based on design intent, amenity, functional design and legislative requirements (CIRIA, 2015; City of Portland, 2016; Melbourne Water, 2005b; Ministry of Housing and Urban-Rural Development (MOHURD), 2014; NIWA, 2012; Public Utilities Board, 2011). However, it is important to identify the critical design factors that are responsible for limiting harmful algal blooms (Melbourne Water, 2005a). Currently, there are only a small number of national standards or codes specifically designed to address HABs in constructed waterbodies. For instance, in the UK, there is no national standard, although due to unique regional hydrology and topography, localized design guidance is recommended during the design and construction phase (Lucas et al., 2014).

Numerous studies have shown that factors such as eutrophication, water temperature, light, and hydrodynamics influence algal bloom formation in natural freshwater waterbodies (Anderson et al., 2002; Paerl & Otten, 2013; Stumpf et al., 2012). Most of these factors also apply to constructed waterbodies. However, current design guidelines for constructed wetlands, lakes and ponds worldwide employ a limited number of design factors to explicitly control algal blooms (CIRIA, 2015; DID Malaysia, 2012; Dorman et al., 2013; Mackay City Council, 2008; Melbourne Water, 2005a, 2017; Ontario Ministry of the Environment, 2003; Public Utilities Board, 2011). Where evident, these criteria generally focus on drivers that occur in the waterbody (e.g., hydraulic residence time or macrophyte cover) or in the adjacent catchment (e.g., impervious area or land use type). The failure to consider a broad range of factors (waterbody and catchment characteristics) may increase the risk of HABs and/or lead to a false sense of design success (where the design process is followed but the waterbody is subject to regular HABs).

Overall, despite the large number of newly constructed shallow waterbodies worldwide, and increasing HAB occurrence, it is unclear whether the available design criteria are appropriate to achieve their design functions while reducing HAB occurrence. To address this gap, in this study, we analyzed design criteria from worldwide guidelines. We then tested the validity of commonly recommended design parameters using a data set of 222 shallow waterbodies in the greater Melbourne region, southeastern Australia, to understand the importance of each factor in predicting HABs. This case study is an opportunity to test the criteria identified in the global guideline review and understand their effectiveness. The contribution of this study is to evaluate the utility and gaps in current design guidelines with respect to HABs. This will help the water managers to develop recommendations and provide useful guidance for future improvements in design guidelines.

2. HABs and Global Constructed Waterbody Guidelines

2.1. Global Guideline Review

For this study, 66 global waterbody guidelines (see Supplementary Information Table S1) were sourced and assessed in relation to their design criteria regarding algal blooms. Approaches and philosophies for stormwater management have a range of names around the world (Radcliffe, 2019) including Water Sensitive Urban Design (WSUD) (Department of Planning and Local Government, 2010; Lewis et al., 2015; Melbourne Water, 2005d), Best Management Practices (BMP) (Pennsylvania DEP, 2006; US EPA, 2004), Low impact development (LID) (Dorman et al., 2013; The City of Edmonton, 2014), Sustainable Urban Drainage Systems (SUDS) (CIRIA, 2015; Martin et al., 2000), Alternative Technologies (Armitage et al., 2013), and Sponge Cities (Ministry of Housing and Urban-Rural Development (MOHURD), 2014).

WSUD was initially proposed in Australia in the 1990s (Allen et al., 2004) and has evolved over time with a number of guidelines produced. Australian WSUD engineering design guidelines (Melbourne Water, 2005d)
have been adopted, modified and applied in similar guidelines in Malaysia (DID Malaysia, 2012), Singapore (Public Utilities Board, 2011), South Africa (Armitage et al., 2013) and China (Ministry of Housing and Urban-Rural Development (MOHURD), 2014). Similar to Australia, LID and BMPs were first used in the USA (ASCE, 1992; Donovan et al., 1995; US EPA, 2004) and spread to Canada (Alberta Environmental Protection, 1999; Ontario Ministry of the Environment, 2003; The City of Edmonton, 2014) and New Zealand (Auckland Regional Council, 2003; Lewis et al., 2015; NIWA, 2012). SUDS was originally developed in the UK (CIRIA, 2007; Ellis et al., 2003; Martin et al., 2000) and other European countries adopted either SUDS, LID or similar techniques (ASTEE, 2017; Conseil général des Hauts-de-Seine, 2007; Vollertsen et al., 2012). Constructed wetlands, retention ponds and lakes are commonly included in these approaches and the technical design sections of the relevant guideline documents were reviewed.

Two methods were adopted to search for relevant guidelines: (a) Review papers on urban constructed waterbodies design guidelines were used to identify relevant search terms (Crowe et al., 2007; Dierkes et al., 2015; Howe et al., 2011; Jayaratne et al., 2010; Lucas et al., 2015; Raspati et al., 2017; Sharma et al., 2018). We then collected and searched design guideline documents and checked if design sections relating to constructed wetlands and lakes were included. (b) We used two internet search engines (Google and Baidu) to search for the keywords ‘WSUD,’ ‘BMP,’ ‘LID,’ ‘SUDS,’ ‘Alternative Technologies,’ and ‘Sponge Cities.’ The gray literature such as industry guidelines is often not available from academic searches such as Scopus, so broader search engines were used. We used these methods to find as many relevant guidelines as possible. As a result we included 66 design guidelines from 15 countries, see Table S1.

Guidelines from Australia and the USA are the most numerous, with each country contributing around a quarter of guidelines reviewed (Figure 1). This is because Australia and the USA were early proponents of more integrated stormwater treatment methods. Although globally there are a range of names, the general underlying design principles are similar. These approaches generally target water quantity and quality issues arising from stormwater, including hydrological design to manage peak flows and water volumes, maintain and improve water quality, and design of stormwater treatment devices (e.g., bioretention systems, swales, wetlands).

This review considers constructed waterbodies that are used for urban stormwater management, which includes wetlands, lakes and ponds. Constructed wetlands are shallow, extensively vegetated waterbodies, which generally have at least two different zones including an inlet zone and macrophyte zones. The water level increases during rainfall events and then slowly reduces to water levels of dry weather, generally over several days. In contrast to wetlands, ponds and lakes have large proportions of open water. They tend to be deeper than wetlands and they have a lower water level fluctuation range. In terms of aquatic vegetation, emergent macrophytes grow in the edges of these waterbodies and submerged plants may exist in the open water. Ponds can be combined with constructed wetlands or are sometimes constructed as ornamental
waterbodies. Lakes and ponds usually have longer residence times than wetlands. Other design values can apply to all three waterbody types.

Overall, the design guideline recommendations are mixed with respect to HABs; some guidelines detail many design constraints (e.g., Melbourne Water (2005c) mentioned five factors related to reducing HABs) whereas others only provide a few specific design values to reduce HABs (e.g., Armitage et al. (2013) mentioned one factor related to HAB reduction). In some cases, the relationships between design factors and algal blooms are only considered indirectly during design or included in the maintenance section of the guidelines. Based on this, the 66 design guidelines have been divided into three categories:

1. Direct: these guidelines explicitly specify design criteria based on reducing or preventing HABs, or refer to another design guideline that directly addresses HABs
2. Indirect: these guidelines do not explicitly include design criteria for HABs but do include criteria that are known to be related to HABs; for example, nutrient concentrations, turbidity, stratification and macrophyte cover, though some guidelines mention regular maintenance and monitoring for HABs. A guideline in this category may also refer to another design guideline that belongs to the indirect category.
3. Not Present: design criteria are based on factors unrelated to HABs (or clearly explain the criteria were designed for another purpose, e.g., safety, flood control, erosion control or mosquito control), and maintenance or monitoring for HABs is not included.

Based on the above categories, 31% of the global guidelines analyzed were in the Direct category and include design criteria focused on reducing HABs. A further 36% of guidelines were in the Indirect category and, to some extent, are beneficial for HAB control. However, around one-third of the global guidelines reviewed did not mention HABs at all.

Despite two-thirds of global guidelines including specific HAB design criteria, there are many constructed waterbodies around the world which frequently experience HABs. This suggests that the existing global design guidelines may not be effective at limiting HABs. This may be because the wrong measures are considered or unsuitable design values have been recommended. For instance, in tropical countries, it may be preferential to include direct design values as their waterbodies may have higher risks of HAB occurrence (i.e., higher temperatures favor cyanobacterial growth) (Paerl & Huisman, 2008). However, based on this review it appears that many guidelines are derived from temperate climate systems with different climate conditions.

### 2.2. Specific Design Criteria Review

In general, there are two types of design factors for constructed waterbodies; those that refer to features of the waterbody itself, here designated as “lake factors,” and criteria that reflect the upstream catchment/watershed that drains to the waterbody, referred to as “catchment factors.”

The aim of many constructed waterbodies is to reduce nutrient levels in the stormwater to protect downstream receiving waters (Ellis et al., 2003; Melbourne Water, 2005d, 2017; National Research Council, 2009). With regards to waterbody design, nutrient reduction can be achieved by physical (e.g., sediment retention) or biological (e.g., vegetation uptake) means (Kadlec, 1989, 2000; Saunders & Kalff, 2001) and other pathways such as chemical transformation and cycling (Fisher & Acreman, 2004; Howard-Williams, 1985). For these mechanisms, the nutrient removal efficiency is influenced by the hydraulic residence times, that is, how long the water remains in the waterbody before being discharged downstream. However, low flushing rates or stagnant water may lead to HABs (Paerl, 2008). Therefore, the design of constructed waterbodies requires a careful balance between enhancing the design efficiency and preventing negative consequences; how small can a waterbody be whilst still reducing HAB occurrence? As inflow, outflow, precipitation and evaporation all influence the hydraulic residence time (Janssen et al., 2014), both lake factors and catchment factors need to be considered when examining the design suitability. Examples of lake and catchment factors are:

1. Lake factors: lake area, depth, aspect ratio, shoreline development index, fetch distance, macrophyte area to lake area ratio, hydraulic residence time.
2. Catchment factors: upstream catchment area, urbanization, imperviousness.
In practice, it is the interaction between multiple factors that affects the overall success of the design (i.e., achieving the waterbody’s design purpose and reduce the risk of HABs). For example, it was found that shallow depths (less than 3 m), high hydraulic efficiency and short hydraulic retention time may provide a robust design in terms of reducing algal blooms in South-Eastern Queensland (Water by Design, 2012), although they would make the waterbody less effective at reducing downstream nutrient loads. The role of single design factors and the interaction between a number of design factors are explored further in Sections 3 and 4.

From the global guidelines review, the 6 most-common design factors from the 66 guidelines relevant to HABs are shown in Table 1. There were 11 design factors (Table S1) found in common across the design guidelines. These factors include HRT, water depth, aspect ratio, hydraulic efficiency, ratio of waterbody area to catchment area, ratio of macrophyte area to waterbody area, SDI, drainage area, hydraulic loading rate, wind, vegetation porosity. For each factor, the percentage of the guidelines that provided advice around that factor was calculated. The top six factors with the highest percentages were then selected for further analyses (summarized in Table S1). Each of these factors is then detailed in the remainder of this section.

The range of design values recommended is explored and the evolution of the recommendations considered. Generally, it was found that the criteria in the guidelines are often focused on ‘Rules of Thumb’ (or empirical values), such as the sizing method for wetlands/lakes (Ellis et al., 2003) or recommendations for aspect ratio (Schueler, 1987). Another common problem was that the sources of the recommended values have not been recently derived (e.g., hydraulic residence time [Hartigan, 1986]). One concerning finding is that, in many cases, current stormwater wetland design guidelines predominantly follow the design process and recommended parameter values from wastewater treatment wetland designs. The source and concentration of pollutants in stormwater runoff are substantially different from wastewater and caution is needed when directly transferring across sectors. Another concern is that many guidelines use similar criteria because the guidelines adopt methods or values from earlier work. This approach does not recognize that criteria may need to be tailored for different climates, hydrology or water quality. These potential limitations are highlighted in subsequent discussions below.

### 2.2.1. Water Depth

Water depth is the most common design parameter for wetlands, lakes and ponds, and it is included in 80% of the reviewed global guidelines. The recommended maximum depths from the guidelines range from 1 to 3 meters. Wetlands have different zones; for example, marsh zones and open water zones, and there are a range of recommendations for the depth of the marsh zones. Marsh zones tend to be shallower than the open water and favor macrophyte growth (CIWIA, 2015; DID Malaysia, 2012; Melbourne Water, 2017; Minnesota Stormwater Steering Committee, 2005). Shallow water depth is essential to ensure efficient mixing throughout the water column (Scheffer, 1997) and sufficient macrophyte establishment (dependent on light penetration) (Scheffer & van Nes, 2007). However, these shallow constructed waterbodies may experience high light penetration and high levels of cyanobacteria in summer when the water is warmer. Deeper water may reduce the resuspension of sediment-bound nutrients, which are potential sources of bioavailable...
phosphorus, into the water column. However, deep lakes are more susceptible to thermal stratification and the subsequent release of nutrients from sediments (Nowlin et al., 2005; Rodusky et al., 2005).

Water flow short-circuiting and anaerobic bottom waters will be reduced when thermal stratification is minimized (Song et al., 2013). Therefore, the lake should be shallow enough to reduce the chances of thermal stratification, but deep enough to minimize the likelihood of algal blooms, which can be triggered or exacerbated by increasing nutrient concentrations in the water column caused by wind-driven resuspension of bottom sediment. The basis for water depth recommendations appears to be Schueler (1987), who recommended that mean depths of 2 m (with over 0.8 ha water surface) may protect against sediment resuspension, although it is noted that stratification may occur in waterbodies shallower than 1 m in some circumstances (Andersen et al., 2017; Vilas et al., 2018). Some guidelines recommend a specific water depth value to achieve adequate residence times for ponds and lakes. To achieve these co-benefits together with other functions, such as safety considerations, the recommended design depth in most guidelines is less than 3 meters. However, the climate and geography of the area also need to be considered. For instance, waterbody characteristics such as icing or melting, stratification and mixing patterns all depend on the climate.

2.2.2. Hydraulic Residence Time

The hydraulic residence time (HRT) (also known as the lake turnover time or detention time) is the third most common global criterion in design guidelines intended to reduce algal blooms, with around 55% of guidelines using this design factor. HRT is defined as the average time that it takes for soluble compounds in water to travel from the inlet of a system to the outlet. The recommended HRT for lakes is usually larger than for wetlands (Department of Water, 2007; Mackay City Council, 2008), as wetlands are generally designed to improve water quality (ensuring adequate treatment) and avoid stagnant water. Therefore, HRTs for constructed lakes and wetlands are further detailed below independently.

For wetland designs, the recommended HRT ranges between 1 to 4 days. The HRT is based on the required reduction rate of suspended particles or nutrients in the wetland. A common minimum value of 24 h appears to be derived from Schueler (1987), who describe the required detention time to achieve an 80% reduction of TSS in urban wastewater wetlands. This value has been widely cited in many design guidelines (Donovan et al., 1995; Ellis, 1992; Ontario Ministry of the Environment, 2003; Silverman, 1988). However, it is not clear if this HRT based on wastewater treatment wetlands is suitable for stormwater design. This is because wastewater wetlands treat a range of source water including municipal and industrial wastewater. These wetlands are generally designed as a tertiary wastewater treatment process and the pollutant composition and concentration in wastewater are different from stormwater-based runoff (Davis, 1995; Kadlec & Wallace, 2008).

For lakes, the HRT design values range from 14-50 days Hartigan (1986) proposed adopting a minimum HRT of 14 days for sizing basins based on total phosphorus (TP) removal rate. This value was adopted from Rast et al. (1983) based on the relationship between algal growth and phosphorus loading in US lakes. It was then adopted more widely (Livingston et al., 1988; NSW EPA, 1997). Other guidelines adopted HRTs to ensure the pollutant concentrations decrease in line with a reduction in either TSS, TP or TN concentrations (St. Johns River Water Management District, 1995). However, the guidelines that use nutrient reduction to manage HABs may need to consider the reduction rate of dissolved forms of nutrients, such as dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP), as algae uptake these bioavailable nutrients more readily for biosynthesis (Aubriot & Bonilla, 2018). A more direct method is to base HRT design criteria on the time it takes cyanobacteria to attain alert-level concentrations. In this case, models (Melbourne Water, 2005c) are used to estimate how long a bloom may take to develop and the HRT is set shorter than this period in order to wash out cyanobacterial cells before they attain bloom-level concentrations. However, this method often assumes that cyanobacterial growth rates in constructed waterbodies are similar to those measured in the laboratory or in small-scale field experiments (Melbourne Water, 2005d). For instance, the current Melbourne Water guidelines in southeastern Australia are based on a small number of in-situ growth rate measurements of Dolichospermum circinale (Melbourne Water, 2005d). Accurate and representative in-situ algal growth rates are difficult to obtain due to methodological limitations (Furnas, 1990; Stolte & Garcés, 2006). Growth rates also differ substantially between and within algal species (Furnas, 1990; Stolte & Garcés, 2006; Willis et al., 2016) and have high temporal variation as conditions in the waterbody change. Relatively few studies have measured in situ growth rates for freshwater bloom-forming
2.2.3. Waterbody Layout: Aspect Ratio and Hydraulic Efficiency

Aspect ratio, which refers to the ratio of the length-to-width (L:W) of waterbodies, is included in almost 70% of the global guidelines. Larger aspect ratios are beneficial to improve sedimentation as well as reducing hydraulic short-circuiting and vertical stratification (Melbourne Water, 2005b, 2005c). It is worth noting that positioning of the lake with respect to the predominant wind direction and the locations of inflow and outflow also influence the hydraulic behavior of the waterbody. Globally, the recommended aspect ratio ranges from 2:1 to 6:1. A minimum aspect ratio of 3:1 was initially adopted based on a general rule of thumb (Schueler, 1987).

Algal scum can accumulate in ‘dead’ pools (zones or spots where the water is stagnant) and, as Griffin et al. (1985) showed for sediment ponds, this dead storage could be reduced by 50% when L:W ≥ 2:1. Later laboratory tests on runoff sampled from sedimentation ponds within a highway construction site considered aspect ratios of 1:1, 2.5:1, and 5:1 to maximize the residence time and increase pollutant removal (Horner et al., 1990). Even though these recommendations were based specifically on tests with highway sedimentation ponds over a limited range of aspect ratios, the results were later cited in a review of 58 monitoring studies in North America (Schueler, 1993) and then adopted by several guidelines (CIRIA, 2007; Department of Primary Industries, 2012; Dorman et al., 2013; Pennsylvania DEP, 2006). This is concerning as it is unclear whether these aspect ratios are appropriate for current design guidelines.

Hydraulic efficiency is typically used in Australian, Singaporean and Swedish design guidelines. It is defined as the ratio of the time of the peak outflow concentration to the nominal HRT to evaluate the effectiveness of a system to achieve its design requirements (Persson et al., 1999). The time of the peak outflow concentration is the time of the peak concentration measured at the outlet of the pond and the nominal detention period is calculated as wetland volume divided by the mean inflow velocity. Hydraulic efficiency measures how effectively the waterbody layout design is in providing equal detention times to all the flows entering the waterbody. Hydraulic efficiency can range from >0 to 1 and higher values indicate a more effective design. Hydraulic efficiencies larger than 0.5 indicate good hydrodynamic conditions (or efficient flushing), with minimal short circuiting and dead zones. A hydraulic efficiency value larger than 0.5 originated from a hydraulic modeling study of 13 hypothetical ponds to represent the influence of pond shape, inlet/outlet locations and types (Persson et al., 1999) and was adopted in Australian runoff quality research (Wong, 2006). Recently, laboratory simulations have been used to optimizing hydraulic efficiency for combinations of different lake shapes, inflow configurations, islands and obstacles (Guzman, Cohen, et al., 2018; Guzman, Nepf, et al., 2018).

2.2.4. Macrophyte Area to Lake Area Ratio

Macrophyte area to lake area ratio (including both submerged and emergent macrophytes) is defined as the ratio of macrophyte area to total surface area within waterbodies. It is included in 45% of all guidelines and the recommended values range substantially from 35% to 85%. Australian guidelines recommend a minimum of 50% areal cover of the lakes and ponds (Melbourne Water, 2005d), while a 35%–65% ratio of vegetation cover to open water is recommended in Canadian guidelines (Government of Alberta, 2018). The macrophyte area to lake area ratio (or abbreviated as macrophyte ratio) varies between wetlands and lakes/ponds, and wetlands have a wider recommended range of macrophyte ratios than lakes. Since there are many different types of constructed wetlands (e.g., wet pond, multiple pond system, wet extended detention pond), the recommended macrophyte ratio for these wetlands is different, which may explain the wide range of recommended macrophyte ratios. The rationale for recommending high macrophyte ratios is because the presence of macrophytes may inhibit cyanobacterial growth (Scheffer, 1997) through different pathways. Macrophytes provide shading that reduce light availability and they compete with algae for nutrients; they have been shown to produce allelopathic chemicals that reduce algal growth rates (Gao et al., 2015; Gopal & Goel, 1993), and they serve as refugia for herbivorous zooplankton that graze on algae (Gebrehiwot et al., 2017; Moss et al., 1998). Therefore, when macrophyte areas within waterbodies are larger, it is less likely that HABs will occur. It has been suggested that the use of macrophytes for algae control...
in temperate zones is more effective than in the (sub)tropics (Jeppesen et al., 2007; Meerhoff et al., 2007); however, other studies report that macrophytes at lower latitudes have similar or greater effects as at higher latitudes (Kosten et al., 2009; Song et al., 2019). Thus, further study is required to determine the optimum macrophyte ratio in different climatic zones. Other factors or activities (e.g., ensuring that plant species are suited to the local conditions, the macrophyte density, and appropriate strategies for harvesting macrophytes including both its frequency and timing) determine the effectiveness of macrophytes in controlling HABs (Kuiper et al., 2017; Zhu et al., 2020). In addition, selecting suitable macrophyte species is also important for improving water quality via the uptake of nutrients (Srivastava et al., 2008). The details will not be discussed here because the recommended species are location-specific.

### 2.2.5. Lake-To-Catchment Area Ratio

Some guidelines provide recommendations on the minimum size of constructed lakes (City of Portland, 2016), although globally the recommended size depends on the local geography and catchment conditions (Ellis et al., 2003; Lim & Lu, 2016). The size of the waterbody relative to its catchment influences the residence time, where a small lake with a large catchment will have a short residence time and vice versa. Therefore, the ratio of the waterbody area to its catchment area is an indirect design factor affecting the likelihood of HABs.

The most common method of sizing the lake surface area is to use an empirical approach that considers the fraction of the connected impervious area that drains into the waterbody and to use a kinetic sizing criterion (known as the Kadlec method) to achieve the required nutrient (TSS, TP and TN) removal target (Kadlec & Wallace, 2008). Approximately 30% of global design guidelines, particularly those released in the 1980 and 1990s, adopt this approach. The recommended waterbody area ranges from 0.2% to 8.0%, however, this approach has been criticized as it may not be reliable for sizing permanent waterbodies (CH2MHILL, 2014; City of Calgary, 2009) and may not apply to current stormwater waterbody design.

A common alternative method is based on simulating the wetland or lake performance to recommend the appropriate lake area as a percentage of its impervious catchment to achieve the nutrient reduction requirements. For instance, the waterbody area is recommended to be at least 3% of the impervious catchment to achieve an 80% reduction for TSS, 45% for TN and TP in Australian WSUD guidelines (Melbourne Water, 2005d). Several other global guidelines have adopted this method and the design value varies due to differences in pollutant removal targets set for each region.

### 2.2.6. Design Criteria Summary

From the review we performed, there are commonalities in the guidelines with the design of constructed waterbodies in relation to their ability to reduce the likelihood of algal blooms. In summary, these are:

1. Design guidance is often based on non-stormwater waterbodies including wastewater treatment ponds or highway sedimentation ponds
2. Empirical or ‘Rule of Thumb’ design criteria are commonly used, and
3. Historical literature (i.e., studies from before the year 2000) is often repeated and potentially used outside of the original context. Few global guidelines have been based on local research

In addition to these concerns, several design factors that may be important for reducing algal growth are not widely included in global design guidelines. For example, the lake shape influences macrophyte growth and wind-driven mixing in the system (Kolada, 2014; Magee & Wu, 2017). The lake shoreline development index can indicate the potential surface area available for macrophyte growth (Janssen et al., 2014). Some guidelines recommend that the waterbody shape should be “irregular,” which is challenging to apply in practice (Auckland Regional Council, 2003; Christchurch City Council, 2003; MDE, 2000). One solution is to use the shoreline development index, but currently this is only explicitly recommended in one recent Canadian guideline (Government of Alberta, 2018).

Lake shape also affects the fetch distance, which is important in controlling wind-driven effects (Tammeorg et al., 2013), such as nutrient resuspension. Up to 90% of total nutrients in a waterbody may be stored in sediments; therefore, wind resuspension needs to be carefully managed (Water by Design, 2012). Minimizing the fetch distance is one way to control for wind in design (Mason et al., 2018), yet none of the reviewed guidelines currently mention this factor. In fact, some guidelines recommend that waterbodies should be
orienteered in the dominant wind direction (Melbourne Water, 2005a) to minimize thermal stratification by wind mixing of the water column. Increasing fetch may reduce the chance of stratification resulting in fewer algal blooms, however, it can also lead to more nutrients released due to wind-resuspension, particularly in shallow waterbodies. Therefore, a careful design for fetch needs further consideration due to these two mechanisms’ opposing effects.

In contrast to implementing individual design parameters, a more holistic approach would be to consider the systemic behavior of constructed waterbodies. A useful paradigm may be found in the literature around systemic changes in natural shallow waterbodies. The theory of regime shifts in shallow lakes considers abrupt changes between clear and turbid water states (Scheffer & van Nes, 2007). Once a waterbody has shifted to a turbid state, it is difficult to reverse to the clear state (Groffman et al., 2006; Scheffer et al., 2001; Vermaire et al., 2017). Assuming that the same behavior applies for constructed waterbodies, it becomes critical to design the waterbody correctly to minimize the likelihood of HABs because, once HABs begin to occur, a new cyanobacteria-dominated regime may be initiated.

This section has focused on commonly used design factors for constructed waterbodies, including several important factors that have been overlooked in most available design guidelines. However, it was hypothesized that no single design factor can clearly explain HAB occurrence patterns. Instead, a risk probability approach was proposed as a useful tool to understand the combined effects of multiple design factors in achieving more resilient constructed shallow waterbodies. In this study, a large data set on constructed waterbody performance was used to examine the suitability of design criteria against HAB occurrence. The relationship between individual design factors and historical HAB data was initially assessed and then the analysis is expanded to consider multiple design factors. Observations of the thresholds that minimize HAB occurrence in the Melbourne region of southeastern Australia were then provided to inform future guideline development.

3. Data and Methodology

To validate the impact of the recommendations from current guidelines, the design criteria outlined above have been assessed against a data set of 222 waterbodies sampled over a period of 9 years. All of these waterbodies are managed by the water utility Melbourne Water, which is located in Victoria, Australia. The greater Melbourne is in southeastern Australia and has a temperate oceanic climate. The average annual temperature in Melbourne is 17°C, and the highest average temperatures are in January and February (26°C). The average annual rainfall varies from more than 500 mm to more than 1,000 mm across the study region. The study area has five large river basin catchments (Werribee, Maribyrnong, Yarra, Dandenong, and Western Port) and it has strong urban growth and high population density.

3.1. Shallow Lakes and Wetlands Data Set

Melbourne Water is a large water utility responsible for managing stormwater within the greater Melbourne area (including Port Phillip and Westernport Catchment, covering an area of approximately 13,000 km²), with oversight of any stormwater waterbodies that have an upstream catchment area of more than 60 ha. Over recent decades, a large number of WSUD waterbodies have been built across the Melbourne Water area of responsibility. Some of these waterbodies were designed and built by Melbourne Water as part of its role in improving water quality in the receiving waters of Port Philip Bay. In other cases, private developers have built stormwater waterbodies, with Melbourne Water responsible for their ongoing maintenance. Melbourne Water has had ongoing concerns relating to HABs in a small number of their constructed waterbodies and, as a result, has undertaken an extensive mapping and cyanobacteria monitoring program for shallow waterbodies within the region.

There are a total of 1,070 constructed waterbodies in the greater Melbourne region. Due to the connectivity between these waterbodies, 222 distinct and separate waterbodies could be identified (Figure 2). In all cases, the waterbodies have water depths less than 3 m. The geographical location (coordinates) and morphological information of the waterbodies were generated from shapefiles (geospatial vectors) provided by Melbourne Water. The geo-locations of all waterbodies were individually verified, with particular care given to the cyanobacteria monitoring sites. The R package ‘LakeMorpho’ was used to obtain the morphological
features of each waterbody, including its area, length, mean width, shoreline development index and the
largest fetch distance (of eight equally spaced fetch directions) based on the shapefiles provided by Mel-
bourne Water. The percentage of the waterbody area covered by macrophytes within each waterbody was
also provided by Melbourne Water. These were estimated from satellite imagery (Fugro, 0.5 m resolution,
captured during December 2016 to Magee & Wu, 2017) and aerial imagery (0.1 m resolution, captured in
January 2017).

Water quality and algal biovolume data were collected and managed under the Melbourne Water cyanobac-
teria monitoring program. Data is available from 2009 to 2018 (Melbourne Water Open Data Hub). Biovol-
ume (in mm$^3$/L) for each cyanobacterial species is calculated using the measured cell density and then mul-
tiplied by the average volume of a cell for that species. The total cyanobacterial biovolume was calculated by
adding the individual biovolume from all cyanobacterial taxa.

Sampling for cyanobacteria was carried out:

1. At regular intervals for sites with frequent historical blooms; and

Figure 2. (a) Average annual rainfall (mm) distribution across the greater Melbourne area from 2009 to 2018 and locations of 222 constructed waterbodies (black dots); (b) average annual temperature (°C) distribution with locations of 222 constructed waterbodies (black dots); (c) urban land use distribution, including urban residential and urban intensive use, with locations of 222 constructed waterbodies (red dots); (d) geographical distribution of waterbodies (black dots) and the log10 median cyanobacterial biovolume (mm$^3$/L) (orange circles) for each sampling site in the study area.
2. On an “as-needed” basis if reports were received from residents or waterbody managers where HABs may have been present

The sampling frequency is once per week for waterbodies for both regular sampling programs and when ‘as-needed’ sampling is carried out. The ‘as-needed’ sampling covered a period of at least one month for each bloom event. Sampling was carried out at a depth of 20 cm below any surface cyanobacterial scum and by choosing the most representative area within the waterbody. In situ water quality data, including pH, electrical conductivity (EC), water temperature, dissolved oxygen (DO) and turbidity, was measured alongside cyanobacterial biovolumes. During the study period, 40 waterbodies (18% of total waterbodies in study area) experienced cyanobacterial blooms, where a bloom is defined as a biovolume \( \geq 4 \text{ mm}^3/\text{L} \) of cyanobacteria (NHMRC, 2008).

### 3.2. Climate and Landscape Data

In this study, local climate data and catchment characteristics were used to analyze the potential drivers of HABs. Gridded climate data, including daily rainfall (5 km resolution), temperature (5 km resolution), solar radiation (5 km resolution) and potential evapotranspiration (PET, 10 km resolution), were obtained from the Australian Bureau of Meteorology (BOM), while wind speed was obtained from CSIRO at a resolution of 1 km (Table 2). These gridded data were generated from station data using two spatial interpolation methods; specifically tri-variate thin plate spline model for the wind data (McVicar, 2011) and Barnes successive-correction method for the remaining variables (Jones et al., 2009). Based on the waterbody shapefiles, daily climate data were extracted for each waterbody from the grids using ArcGIS. This 9-year daily data set was then averaged over the study period so that a single value for each weather variable could be used in the categorical analysis.

Elevation data was obtained from a Digital Elevation Model (DEM) with 25 m horizontal resolution from Geoscience Australia (GA) (Table 2). The upstream catchment for each waterbody was delineated by using the Archydro function in ArcGIS 10.6. The stream order for the stream/drainage line directly upstream of each waterbody was also generated.

Australian Catchment Land Use data with 50 m resolution were downloaded from the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES). The land use is based on the Australian Land Use and Management (ALUM) Classification version 8, and the urban land use (urban residential and urban intensive) was extracted for the upstream catchment for each waterbody. The catchment imperviousness data were provided by Melbourne Water and were extracted for each upstream catchment (Table 2).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
<th>Counts/Resolution</th>
<th>Dates</th>
</tr>
</thead>
<tbody>
<tr>
<td>Biovolume</td>
<td>mm(^3)/L</td>
<td>63 sites</td>
<td>2009–2018 (non-continuous)</td>
</tr>
<tr>
<td>Climate data</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rainfall</td>
<td>mm</td>
<td>0.05°</td>
<td>2009–2018</td>
</tr>
<tr>
<td>Temperature</td>
<td>°C</td>
<td>0.05°</td>
<td>2009–2018</td>
</tr>
<tr>
<td>Potential Evapotranspiration (PET)</td>
<td>mm</td>
<td>0.05°</td>
<td>2009–2018</td>
</tr>
<tr>
<td>Solar radiant exposure</td>
<td>MJ/m(^2)</td>
<td>0.05°</td>
<td>2009–2018</td>
</tr>
<tr>
<td>Wind</td>
<td>m/s</td>
<td>0.1°</td>
<td>2009–2018</td>
</tr>
<tr>
<td>Catchment and waterbody data</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Waterbody morphology</td>
<td>n.a.</td>
<td>222 waterbodies</td>
<td>2018</td>
</tr>
<tr>
<td>Digital Elevation Model (DEM)</td>
<td>m</td>
<td>25 m</td>
<td>n.a.</td>
</tr>
<tr>
<td>Catchment land use</td>
<td>n.a.</td>
<td>50 m</td>
<td>2018</td>
</tr>
<tr>
<td>Directly Connected Impervious (DCI)</td>
<td>%</td>
<td>Entire region</td>
<td>2013</td>
</tr>
<tr>
<td>Percentage of macrophyte cover in each waterbody</td>
<td>%</td>
<td>1,070 sites</td>
<td>2018</td>
</tr>
</tbody>
</table>

Table 2: Summary of Cyanobacteria Data, Climate and Landscape Data
3.3. Analysis Methods

The aim of the data analysis was to understand if the occurrence of HABs can be explained by reference to the design factors for constructed waterbodies. This Melbourne Water data set is an ideal data set to test the influence of design factors as it contains a large array of size, shape, catchment size, and related parameters over a large area with variable climate and levels of urbanization. Preliminary data screening showed no strong relationships between biovolumes and any one individual design factor (see Supporting Information S1). This result is consistent with similar analysis of 46 lakes in Western Australia (ENV Aus, 2008). Therefore, the focus of the analyses was on explaining bloom occurrence. Bloom occurrence was considered for each factor individually as well as considering the influence of multiple factors together.

K-Nearest Neighbor (KNN) classification was used to understand the relationship between multiple design factors and bloom occurrence.

3.3.1. Single Factor Analysis

A single factor analysis was undertaken by comparing the empirical cumulative distribution functions of a specific design factor for bloom and non-bloom waterbodies. It was assumed that, if a design variable was important in explaining HABs occurrence, then the distribution of values for that variable would differ from the distribution for non-bloom waterbodies. The single factors were tested include aspect ratio, ratio of macrophyte area and waterbody area, ratio of waterbody area and urban area, ratio of waterbody area and impervious area, fetch, SDI. Due to the lack of available data, other factors included in Table 1 were not tested.

Mathematically, it was assumed that there are \( n \) design variables that are represented as \( F_i \) (\( i = 1, 2, \ldots, n \)). The blooming threshold was set as cyanobacterial biovolume \( \geq 4 \text{ mm}^3/\text{L} \), based on the Australian Level 1 HAB warning (NHMRC, 2008). If there are \( m \) waterbodies that have experienced at least one bloom during the study period, then the \( m \) values of each factor are ranked (one from each waterbody) from largest to smallest (Equation 1).

\[
F_i \rightarrow \begin{cases} 
F_{i,j} & (i = 1, 2, \ldots, n; j = 1, 2, \ldots, m) 
\end{cases}
\]

The empirical cumulative distribution function (CDF) was constructed by calculating how many waterbodies have design values greater than some threshold value. The cumulative probability \( P_{ij} \) is the ratio of the number of waterbodies \( (N_{ij}) \), with \( F_i \geq F_{ij} \) the total number of waterbodies that have experienced at least one HABs \( (N_t = m) \) (Equation 2).

\[
P_{ij} = \frac{N_{ij}}{N_t} 1 \left\{ \{F_i \} \geq \{F_{ij}\}, i = 1, 2, \ldots, n, j = 1, 2, \ldots, m \right\}
\]

This analysis was repeated for waterbodies that have never recorded a bloom. The two CDFs were then compared using a Kolmogorov Smirnov two sample test (Lehmann & Romano, 2006) to establish if there are significant differences in the distributions of any particular design factor for bloom and non-bloom waterbodies.

3.3.2. Multiple Factor Analysis

Multiple design factors will likely interact with each other to potentially enhance or reduce the probability of a bloom. To compare multiple design factors (the same factors tested in the single factor analysis), KNN was used to identify important combinations of design factors that may lead to algal blooms. KNN is a non-parametric method commonly used in classification problems (Hastie et al., 2009).

The KNN method predicts \( \hat{y} \), by averaging the values \( y_i \) from their nearest neighbors \( (N_k(x)) \), defined by the \( k \) closest points \( x_i \) in the training set, using their Euclidean distance. Since \( y_i \) represents categorical data, the predicted values \( \hat{y} \) are rounded to 0 or 1 to classify blooms or non-blooms respectively.
Five-fold cross validation was used to estimate the optimum number of neighbors (k) with potential values for k tested between 1 and 10. The five groups were selected using stratified sampling to ensure the number of waterbodies with blooms was approximately the same in each cross-validation subset. The optimum number of neighbors was selected by considering the Equitable Threat Score (ETS) and the Heidke Skill Score (HSS), which measure the skill of a forecast in terms of correctly modeled occurrences and non-occurrences compared to the total number of predictions. The ETS considers the skill of the model compared to the random chance of predicting a bloom, whilst the HSS compares the skill of the model compared to the skill of a model that randomly predicts both blooms and non-blooms. After selecting the optimum number of neighbors, the full data set was used to explore the likelihood of blooms across a sample space of multiple design factors and to suggest joint thresholds that may be useful for future investigations and design guidelines.

4. Results

The results for single and multiple design factor analyses are summarized in the text below and compared to the recommended design factor values (Table 1). The analyses are based on 222 waterbodies and 35 of these waterbodies have had at least one HAB in the 9-year study period. The design factors from Table 1 that were tested are: aspect ratio, macrophyte area to lake area ratio, lake to catchment ratio. In addition to these, fetch and shoreline development index were tested because (a) they may be influential factors based on the reviewed literature and (b) both wind and shoreline geometry are qualitatively described in many guidelines.

4.1. Constructed Waterbodies and HAB Statistical Analyses

Characteristics of 222 constructed waterbodies were analyzed in the greater Melbourne region. Waterbodies in the east receive more rainfall than western ones, with a rainfall gradient of 500 mm to 1,000 mm over the study region (Figure 2a). Annual average temperatures are between 17°C and 20°C (Figure 2b). Most of the waterbodies have been constructed within urbanized or residential areas and nearly all of the waterbodies that have experienced HABs are in highly urbanized catchments (Figures 2c and 2d). The total biovolume medians for waterbodies in highly urbanized areas are higher than in other areas (Figure 2d).

It is possible that recently constructed waterbodies might not have as many blooms if design guidelines have improved over time. Increasing age of constructed waterbodies may also increase the likelihood of HABs due to nutrient-rich sediment accumulation. Figure 3a illustrates the impact of the waterbody construction date (waterbody age) on HABs, noting that some waterbodies may have experienced modification or silt removal throughout time. Grouping waterbodies built at the same time shows no change in the likelihood of HAB occurrence over the study period, which could be expected if design guideline had improved over time (Figure 3a). Higher biovolumes tend to occur during the Australian summer (December to February), when water temperatures are higher (Figure 3b).

Due to the implementation of the Melbourne Water monitoring program, most of the waterbodies were sampled at only one location. Therefore, the in-lake spatial variability was not considered. Since the aim of this study was to understand the differences in HABs occurrence due to waterbody design differences, the representative biovolume of each waterbody was calculated as the temporal median of all recorded cyanobacterial biovolumes for that location. For waterbodies with more than one sampling site, the biovolume was calculated as the average of all sampling sites within the waterbody. The sampling methods for Melbourne data contain both regular sampling for some waterbodies and on an as-needed basis for other waterbodies. Because of this sampling strategy, HAB biovolumes from the ‘as-needed’ samples were typically higher than for sites with temporally regular sampling, because the former approach was biased to bloom periods while the latter approach collected data during both bloom and non-bloom periods. However, given
the aim of the study is to understand the factors that affect whether a waterbody experiences a bloom or not, the different sampling methods are not expected to cause any biases in the analyses.

4.2. Influence of Single Design Factors

Empirical CDFs have been used to understand whether there are differences in the distributions of each parameter for bloom and non-bloom waterbodies (Figure 4). Significant differences are evident in the distributions for the macrophyte ratio, fetch distance and SDI; which are different at a 5% level based on a two sample Kolmogorov-Smirnov test. For the aspect ratio, the ratio between waterbody area to urban area, and the ratio of waterbody area to impervious area, there are no significant differences in the distributions between bloom and non-bloom waterbodies.

Waterbodies that have not experienced blooms have higher proportions of macrophyte cover than bloom waterbodies (Figure 4b). This is consistent with expectations that increased vegetation cover will reduce nutrient loads, while also limiting cyanobacterial growth (Scheffer, 1997). Internationally, a wide range of macrophyte cover is recommended (25%–85%), with 50% macrophyte cover recommended by local guidelines (Melbourne Water, 2005a). Maximizing macrophyte cover is an important recommendation based on these results, but it is evident from Figure 4b that blooms can occur even when macrophyte cover is high. This suggests that high vegetation rates cannot completely protect against other design or maintenance issues or that macrophyte type (not just percentage) may also be an important design factor. However, it is important to note that macrophyte area can be highly variable both spatially and temporally. Given that the macrophyte area was calculated based on a single point in time, it may be that the HABs occurred when the macrophyte cover was less than the value assigned to the waterbody.

Fetch is another factor where significant differences are observed between bloom and non-bloom waterbodies (Figure 4e). Interestingly, fetch has not been included in any of the global guidelines that were reviewed, although it is known that wind effects are important in terms of nutrient resuspension (Blottière et al., 2014; Carper & Bachmann, 1984; Schelske et al., 1995). Moreover, strong winds can damage the macrophyte beds and potentially increase the likelihood of HABs (Havens et al., 2016). Minimizing fetch to minimize nutrient resuspension should contribute to lower nutrient levels and fewer HABs. It should be noted that large fetch can sometimes improve mixing rates within waterbodies and prevent thermal stratification, which would reduce the likelihood of HABs.

Shoreline Development Index (SDI) describes the irregularity of waterbodies and the potential surface area available for macrophyte growth. Currently, SDI is only mentioned in a single guideline (Government of...
which recommends a value greater than 1.2. Figure 4f confirms that larger SDIs tend to reduce HABs risk, although there appears to be an upper limiting value (around 3) above which there are no further benefits. The interaction of SDI with other design factors may explain this limit and the following section explores two- and three-factor relationships.

Aspect ratio was one of the most common design factors recommended in the design guidelines reviewed. However, the results (Figure 4a) suggest that aspect ratio alone cannot explain the differences between bloom and non-bloom waterbodies in the Melbourne Water data set. Australian guidelines recommend that the aspect ratio should be at least 5:1, and the global guidelines have a criteria range between 2:1 and 6:1. Almost 70% of Melbourne Water's waterbodies with blooms meet these criteria, yet many still experience algal blooms. Similarly, a large proportion of bloom waterbodies follow recommendations for the ratio of the waterbody area to upstream urban area (Figure 4c) and the ratio of the waterbody area to upstream impervious area (Figure 4d), but these criteria have little impact on HABs in Melbourne's constructed waterbodies. While these three factors may be important for other water quality objectives, such as stormwater management, our findings suggest that these factors are not as important as the macrophyte area, SDI or fetch in terms of future investigations and guidelines for reducing HABs.

4.3. Multiple Factor Analysis

Combinations of multiple factors were considered to explore whether interactions between different design factors can be identified. For two-factor analysis, the KNN method was used to qualitatively understand important interactions between design factors and to identify potential design thresholds. Based on a 5-fold...
cross validation, the optimum number of neighbors was found to be between 5 and 7, depending on which combination of factors was tested; hence, 7 neighbors were adopted for the analyses. The ETS and HSS results for each of the KNN models are shown in Supplementary Information Table S2. ETS and HSS values larger than zero indicate that the model has better skill than random and higher values indicate better prediction skill of the model.

To illustrate the value of considering multiple design factors, Figure 5 presents three cases from the two-factor analysis. Based on the results of one-factor analysis, the macrophyte cover, fetch and SDI were found to be useful in separating HABs occurrence versus non-occurrence (Figure 4). These three factors were then combined to identify any mutual thresholds that might be useful for design. Given the importance placed on aspect ratio in the design guidelines, it was considered in combination with macrophyte area. Other two-factor combinations are provided in the Supporting Information S1.

The KNN model for macrophyte area and SDI had some modest skill, with an ETS of 0.14, and a HSS of 0.20 (Supplementary Information Table S2). The KNN model suggests that, when the macrophyte area is less than 40% and the SDI is less than 1.5, waterbodies have a higher likelihood of experiencing blooms (Figure 5a). Similarly, low SDI values combined with a large fetch also increase the risk of HABs (Figure 5b). However, some bloom waterbodies that comply with these design criteria still experience blooms, suggesting that there are other factors influencing bloom occurrence. Importantly, the KNN models are not intended to develop recommendations for design criteria, but, instead, we argue that these thresholds may provide guidance for future process-based investigations.

Figure 5c highlights an example of where KNN modeling was unable to detect a relationship between bloom occurrence and the design factors under consideration. When macrophyte area is combined with aspect ratio, the model has almost no skill (ETS = 0.06) and no regions of this multifactorial space have been identified to have a high likelihood of supporting blooms. This confirms the results from the one factor analysis that aspect ratio does not provide a good differentiation between bloom and non-bloom waterbodies.

Finally, the KNN method was used to consider three co-design factors. In this case, the three most influential individual factors (macrophyte area, SDI and fetch) were modeled using KNN (Figure 6). The optimum number of neighbors was again found to be 7 through cross validation. To illustrate the multifactorial space that increases the likelihood of blooms in three dimensions, the KNN exposure space was sliced into three cross sections at the 5th, 50th and 95th percentiles of fetch. When fetch is small (5th percentile), there are no areas in the multifactorial space that flag increased risk of HABs. As fetch increases to its median value (around 150 m), some areas of increased HAB risk are predicted for low SDI values. Finally, if the fetch is very long, a high risk of HABs is flagged for low SDI (less than 2) and low macrophyte cover (less than 60%). As with the two-factor analyses, these thresholds would provide useful starting points for process-based studies of algal blooms. The addition of fetch to the macrophyte area/SDI analysis suggests that, if the fetch...
of the waterbody is very long, higher macrophyte areas and/or higher SDI values are required to offset the impact of fetch on HABs.

5. Discussion

In this study, 66 global design guidelines for constructed shallow waterbodies were reviewed. Only one-third of global guidelines directly mention algal bloom control measures with the remainder indirectly designing for blooms or not considering algal blooms at all. The review found that there are six commonly used design factors that aim to reduce/prevent algal blooms, including hydraulic residence time, water depth, aspect ratio, percentage of waterbody area to drainage area, macrophyte ratio within waterbodies and hydraulic efficiency. The recommended values for these factors range widely and it was shown that the values are often based on a single publication and, in some cases, values are based on studies of non-stormwater systems.

To further investigate the influence of engineering design on HAB occurrence, a large data set of constructed waterbodies with HAB monitoring over 9 years was analyzed against design parameters, including aspect ratio, macrophyte area to waterbody area ratio, waterbody area to catchment area ratio, fetch and shoreline development index. The data set is valuable as it spans a large spatial and temporal range, but it is worth noting that limited data is available on water depth, hydraulic residence time and hydraulic efficiency. Local advice suggests that the average water depth for the shallow, constructed waterbodies analyzed in this study is less than 3 meters. The hydraulic residence time is not available as flow into and out of stormwater waterbodies is rarely gauged. However, given that the water depths are assumed to be similar for all waterbodies, then the hydraulic residence time should be proportional to the ratio of the lake area to the upstream catchment area. Finally, hydraulic efficiency can be partially represented by the aspect ratio.

Our analysis based on the Melbourne Water data set indicates that macrophyte ratio, fetch and shoreline development index (SDI) are the most influential single design factors for HABs in these constructed waterbodies. Specifically, HABs often occur in shallow constructed waterbodies with low macrophyte cover, long fetch distances and/or low shoreline irregularity. In contrast, aspect ratio, lake/urban ratio or lake/impervious ratio do not substantially influence HABs individually, but they may be important for other design purposes. The macrophyte ratio combined with SDI, and SDI combined with fetch, are the two-factor combinations that best separate bloom from non-bloom waterbodies. When the macrophyte ratio is less than 40% and SDI is lower than 1.5, or when SDI is lower than 1.8 and fetch is shorter than 200 m, there is a higher risk of HAB occurrence in these waterbodies. For three-factor combinations, only the macrophyte ratio combined with SDI and fetch length provides additional insights compared with the two-factor analysis. These results indicate that it is necessary to increase both macrophyte ratio and SDI if a waterbody has a very long fetch distance. This analysis also suggests that large fetch can also produce negative outcomes, and that improved mixing should be achieved by other means to suppress HABs, such as artificial mixing.
Above suggest that most global criteria are similar (see Section 4.2). Future work should consider the temporal and spatial variability of macrophyte area ratios.

While KNN models indicate some skill in differentiating between bloom and non-bloom waterbodies, many bloom waterbodies outside the high-risk multifactorial regions still have blooming events. Our KNN model results can guide priorities for future investigations such as process-based studies that can inform waterbody design guidelines. However, our study suggests that even when recommended design values are achieved, algal blooms can still occur in some waterbodies. This suggests that overarching global guideline values that have been applied widely are likely to be flawed and that individual detailed designs are required on a case-by-case basis. This situation may arise because engineers and managers may use existing guidelines as a “cookbook,” without site-specific context. In general, design guidelines tend to stifle innovation and emphasize ‘cookie-cutter’ design (Bradford & Gharabaghi, 2004); however, we note that is, better to have some guidelines than none, which would promote reliance on “rules of thumb.”

Of even more concern is the use of guidelines and criteria developed for one specific location in different jurisdictions or even countries. For instance, guidelines in Singapore (Public Utilities Board, 2011), Malaysia (DID Malaysia, 2012), South Africa (Armitage et al., 2013), and some trials in China follow the design processes and criteria of Australian WSUD guidelines (Melbourne Water, 2005d), as described in Section 2. The same guidelines also recommend using the Australian-derived MUSIC model (Model for Urban Stormwater Improvement Conceptualization) (eWater, 2012) for conceptual design of waterbodies. The results above suggest that most global criteria are similar (see Section 2 and specifically Table 1), but that design values and management need to be tailored for individual sites, even within local regions (e.g., Melbourne Water’s region of responsibility).

If guidelines continue to be developed and utilized, then it is vital that they move toward a more direct and process-based design for preventing HABs. Indirect design guidelines are typically based on water quality objectives, such as reducing TN and TP, which may or may not have an impact on HABs. While decreased nutrient loads are associated with a reduced likelihood of HABs, the design process may need to focus on controlling bioavailable dissolved nutrient forms such as DIN and SRP (Paerl, 2014). Moreover, different localities may need to emphasize different tools for HABs control in their guidelines. For instance, for countries in tropical climate zones where the risk of cyanobacterial blooms is higher, it is more important for local guidelines relating to waterbody design to be implemented. Given the potential for increased proliferation of algal blooms under global climate change (Ho et al., 2019; Huisman et al., 2018; Mooij et al., 2007; Paerl & Huisman, 2008; Visser et al., 2016), it is crucial that local design guidelines are increasingly directed toward controlling HABs.

Design criteria and processes for constructed waterbodies are a trade-off between multiple objectives. For example, the purpose of many constructed wetlands is to reduce downstream nutrient loads to protect sensitive ecosystems (i.e., downstream receiving waters) from eutrophication (Jayaratne et al., 2010; Melbourne Water, 2005b). This means that the wetland itself is a sink for those nutrients, which can lead to eutrophication in the constructed waterbody for the benefit of downstream environments. Therefore, specific design criteria that prevent HABs may also lead to degradation of other functions or purposes of the wetlands (Bledsoe et al., 2020). For example, short residence times can reduce the possibility of HABs (Ho & Michalak, 2020; Reichwaldt & Ghadouani, 2012), but longer residence times are required to remove nutrients and achieve overall nutrient reduction targets in the catchment. In these cases, the design trade-offs may be approached with a risk-based framework where the design criteria selected are focused on the downstream risks.

It was evident when reviewing guidelines that have been revised and updated, that the design factors related to HABs have evolved over time. This demonstrates how guidelines could be updated as the impacts of climate change and their uncertainty are better quantified, as well as including new research on algal blooms in stormwater waterbodies. These changes in design guidelines over time may shed light on how understanding of HABs has evolved. Interestingly, there was little difference in algal biovolumes measured in old waterbodies (constructed before WSUD guidelines were released in 2005) and newer waterbodies in the Melbourne Water data set (Figure 3a). However, we note that very few ‘bloom’ waterbodies have been
constructed since 2005. There are three potential reasons for this: (a) many of the older waterbodies have been redeveloped (including reconstruction, vegetation regeneration and water depth control) following updated design guidelines; and (b) meeting the current design guidelines does not necessarily reduce the likelihood of HABs; and (c) there is often a delay in blooming onset while nutrients, especially phosphorus, accumulate in the waterbody.

Integrating both lake factors (e.g., aspect ratio, SDI, fetch distance, macrophyte area to waterbody area ratio) and catchment factors (e.g., waterbody area to urban area ratio) when designing waterbodies to reduce HABs risk is essential. Our multi-factor analysis suggests that appropriate design guidelines for the macrophyte ratio and the size of the waterbody (as a proportion of the upstream impervious area) can be combined to reduce the likelihood of HABs. However, it should be acknowledged that some design factors are impossible or unrealistic to change in practice. When developing a newly constructed waterbody within an existing urbanized catchment, or retrofitting an existing waterbody, factors (such as its location, upstream land use and size of the upstream catchment) are partially or completely constrained, such as its location and the land use and size of the upstream catchment. Our multiple factor analysis identifies which design factors can be optimized given these constraints. For example, the macrophyte ratio, aspect ratio or SDI of the waterbody could be increased.

Appropriate HABs design is critical because the theory of regime shifts (also known as alternate stable states) indicates that it is difficult to reverse a cyanobacteria-dominated state to a non-HAB state (Cottingham et al., 2015; Scheffer et al., 2001). Once large scale HABs occur in constructed waterbodies, considerable resources are required to ameliorate these problems. Worldwide, efforts are underway to control and reduce HABs (Paerl & Otten, 2013), including physical treatments such as surface scum removal, vertical mixing, enhancing flushing, reducing water levels, and chemical treatment such as the addition of algaecides and sorbent minerals (Bullerjahn et al., 2016). However, the number of proven mitigation technologies (physical and chemical treatments) is limited (Paerl et al., 2016; Stroom & Kardinaal, 2016), and their applicability and efficacy rely on other variables, such as weather conditions (Bullerjahn et al., 2016; Paerl, 2018). Regular maintenance of constructed waterbodies is also important for minimizing HABs and reducing the need for mitigation measures. In most design guidelines, the recommended maintenance methods for HABs control are limited, with the maintenance of macrophytes the most-commonly recommended technique (City of Portland, 2016; Melbourne Water, 2005c; NIWA, 2012). Another alternative is to use algal dispersion strategies (such as artificial mixing and surface scum removal) once an algal bloom is observed (Paerl et al., 2016; Visser et al., 2015). It is also recommended that silt is removed every 5–10 years if frequent algal blooms occur (Melbourne Water, 2005a). Sediment removal can help to remove nutrients from waterbodies.

The large Melbourne Water data set on algal blooms in constructed waterbodies has allowed the development of KNN-based method to identify important design factors for these systems. The methodology used in this study can be applied globally. However, it is important to note that specific findings of the key factors and their combinations are specific to the Melbourne Water data set and their wider applicability needs to be tested. This is because of the differences in hydrological patterns, climate, pollutant reduction targets and/or allowable nutrient levels in different jurisdictions and ecological systems. Despite the relatively wide average rainfall and temperature distributions, our findings are likely most relevant to similar temperate climatic zones, with winter dominated rainfall regimes. Therefore, it is recommended that similar analyses are carried out in other parts of the world with different geographical and climatic zones. Constructed shallow waterbodies in tropical and subtropical regions can experience long thermal stratification (Song et al., 2013). Therefore, detailed hydrodynamics modeling studies (Hodges, 2014) are needed in guidelines design process to understand the importance of mixing and stratification in shallow constructed waterbodies.

One limitation of this study is the restricted amount of field data for some sites. Since the Melbourne Water sampling program uses a combination of regular sampling for high-risk waterbodies and community reporting for low-risk waterbodies, it is likely that some blooms may have been missed. Another limitation is the location of the sampling. Cyanobacterial biovolumes sampled from the waterbody center and edge may be different because the cyanobacterial distribution can be spatially variable. One promising future strategy is to use remotely sensed data to complement in-situ data in sparsely sampled areas. Remote sensing instruments, such as the Medium Resolution Imaging Spectrometer (MERIS) and the Moderate Resolution Image Spectrometer (MODIS), have been applied to identify bloom magnitude in large lakes (Clark et al., 2017;
Karki et al., 2018; Lunetta et al., 2015; Urquhart et al., 2017) for many years and algal bloom detection algorithms are well developed and widely applied in the USA (Lunetta et al., 2015). However, the resolution of these products is insufficient for small urban waterbodies. Recently, satellite products with higher resolution, such as Landsat and Sentinel, are increasingly being used to identify harmful algal blooms (Clark et al., 2017; Malthus et al., 2019; Toming et al., 2016). Algorithms have been developed for both coastal and inland freshwaters (Ho et al., 2017). Using these techniques to compliment the in-situ field data may be a powerful way to re-examine the criteria in current design guidelines.

6. Conclusion

In this study, we reviewed global design guidelines from WSUD, BMP, LID, SUDS, and other publications on constructed shallow lakes and wetlands. Nearly one-third of these guidelines describe design values that are directly related to HAB control. These direct design factors will be increasingly important in new or updated guidelines due to the increased risk of HAB occurrence and severity as a result of climate change. The most common design factors in current guidelines include hydraulic retention time, water depth, aspect ratio, hydraulic efficiency, waterbody/urban ratio, and macrophyte ratio. However, upon researching the source reference material of these guidelines, it was evident that many guidelines are based on non-stormwater waterbodies, empirical or 'Rule of Thumb' design criteria, and dated literature. Moreover, some design factors that are known to be important for HAB control are rarely mentioned in the guidelines. For instance, variations in wind fetch distance and shoreline development index (SDI) are important design criteria but they are not listed in existing global guidelines.

To assess the suitability of current design factors, analysis of the likelihood of HABs was conducted on a large number of constructed shallow urban waterbodies in the greater Melbourne region in southeastern Australia. The one-factor analysis results indicate that macrophyte ratio, SDI and fetch distance are the most influential factors in reducing HABs within this study area. Macrophyte ratio combined with SDI, and SDI combined with fetch, are the most influential two-factor combinations, whereas macrophyte ratio, SDI and fetch form the strongest three-factor predictor of the likelihood of algal blooms. These results may provide guidance to reduce HAB occurrence in future design guidelines updates.

Design guidelines need an improved focus on process-based studies and in-situ data to inform their recommendations. We have developed a method that can identify critical factors which should be prioritized in future design guidelines development, some of which have not been included to date in such guidelines. Considering these factors based on the review and the test on the studied constructed waterbodies should form a major improvement in the guidelines to reduce the HAB occurrence.

Future studies are recommended to focus on the following research questions to improve our current knowledge of design guidance to control HABs in constructed shallow waterbodies:

1. Apply this methodology to constructed waterbodies in other regions, with different climate and hydrology
2. Can remote sensing measurements supplement field data to increase the spatial and temporal coverage of potential HABs?
3. Is it possible to reliably measure in-situ growth rates for more freshwater bloom-forming cyanobacteria under different hydraulic residence times?

Data Availability Statement

The data archiving is underway and the data will be available from Melbourne Water Open Data Hub (https://data-melbournewater.opendata.arcgis.com/search?collection=Dataset&tags=drainage). The data to be archived has been included in the Supporting Information for review purposes.

References


Acknowledgements

The authors wish to acknowledge the ongoing support of Melbourne Water via the Nuisance and Harmful Algae Science-Practice Partnership (NHASP) for this research project. Ms Shuang Liu is supported by NHASP and an Australian Government Research Training Program scholarship.


Havens, K., Paerl, H. W., Philips, E., Zhu, M., Beaver, J., & Srifa, A. (2016). Extreme weather events and climate variability provide a lens to how shallow lakes may respond to climate change. *Water, 8(6), 229. https://doi.org/10.3390/w8060229


St Johns River Water Management District. Environmental resource permits: Regulation of stormwater management systems.
Author/s:
Liu, S.; Johnson, F.; Tamburic, B.; Crosbie, N. D.; Glamore, W.

Title:
The Effectiveness of Global Constructed Shallow Waterbody Design Guidelines to Limit Harmful Algal Blooms

Date:
2021-08

Citation:

Persistent Link:
http://hdl.handle.net/11343/289524

License:
CC BY