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Published

2010

Journal Title

Water Research

DOI

10.1016/j.watres.2010.06.016

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Predicting the vulnerability of reservoirs to poor water quality and cyanobacterial blooms

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Abstract

Cyanobacterial blooms in drinking water reservoirs present a major ecosystem functioning and human health issue. The ability to predict reservoir vulnerability to these blooms would provide information critical for decision making, hazard prevention and management. We developed a new, comparative index of vulnerability based on simple measures of reservoir and catchment characteristics, rather than water quality data, which were instead used to test the index's effectiveness. Testing was based on water quality data collected over a number of seasons and years from 15 drinking water reservoirs in subtropical, southeast Queensland. The index correlated significantly and strongly with algal cell densities, including potentially toxic cyanobacteria, as well as with the proportions of cyanobacteria in summer months. The index also performed better than each of the measures of reservoir and catchment characteristics alone, and as such, was able to encapsulate the physical characteristics of subtropical reservoirs, and their catchments, into an effective indicator of the vulnerability to summer blooms. This was further demonstrated by calculating the index for a new reservoir to be built within the study region. Under planned dimensions and land use, a comparatively high level of vulnerability was reached within a few years. However, the index score and the number of years taken to reach a similar level of vulnerability could be reduced simply by decreasing the percentage of grazing land cover via revegetation within the catchment. With climate change, continued river impoundment and the growing demand for potable water, our index has potential decision making benefits when planning future reservoirs to reduce their vulnerability to cyanobacterial blooms.

Key Words

Agricultural land use, algae, climate change, decision making, eutrophication, watershed

1 Introduction

In the tropics and subtropics, climate and the physical characteristics of reservoirs create conditions that promote algal blooms (Jones and Poplawski 1998). Higher air temperatures and longer daylight hours in summer months lead to stronger thermal stratification and, potentially, the release of bioavailable nutrients from anoxic sediments (Burford and O'Donohue 2006). Summer-dominated rainfall and subsequent inflows also tend to increase nutrient supply during this time. Together with the long water residence times typically associated with reservoirs, as opposed to river systems, these factors can combine with other, complex causative factors to make summer blooms a common phenomenon (McGregor and Fabbro 2000). Indeed, there is growing concern over the increased reporting of toxic cyanobacterial blooms in reservoirs the world over, particularly during summer months (e.g Padisák 1997; Bouvy et al. 2000; Wiedner et al. 2007). In addition, global warming is predicted to increase nutrient loads and algal growth in temperate systems (Matzinger et al. 2007; Paerl and Huisman 2008).

Catchment (= watershed) characteristics are also associated with changes in water quality and algal blooms within impounded systems. In particular, agricultural land use can lead to reservoir eutrophication: 1) by increasing soil erosion and nutrient loads in runoff, which can cause changes in nitrogen and phosphorus ratios; or 2) through a combined effect with other catchment and reservoir characteristics, such as water storage capacity and catchment size (Arbuckle and Downing 2001; Knoll et al.

2003; Davis and Koop 2006; Yang et al. 2008). Because summer conditions favour algal blooms and, particularly in the tropics and subtropics, tend to occur in tandem with increased precipitation (which increases the inflow of nutrients and sediments), the effects of land use on reservoir water quality may be most apparent during warmer months.

In response to the increased demand for irrigation and drinking water supply by growing human populations, new reservoirs are being built and/or the storage capacities of existing reservoirs increased (L'Vovich 1990; Pringle 2001; Zengi et al. 2007). In developing countries, particularly China and India, new reservoirs in both urban and remote areas continue to be constructed (Dudgeon 2000). Much of this reservoir expansion is occurring in the tropics and subtropics, and often in conjunction with conversion of forest or savannah into agricultural lands (Blanch 2008; Gücker et al. 2009). This combination of factors suggests that the incidence of algal blooms will become more likely. In turn, there is a need to reliably forecast the occurrence and frequency of toxic algal blooms in existing or future reservoirs.

Regular monitoring within drinking water reservoirs is conducted to ensure that the relevant water quality guidelines for ecosystem and human health are met. This process, and the supply of safe drinking water, however, consumes considerable human and fiscal resources (De Ceballos et al. 1998). In addition, monitoring often commences after reservoirs are built and water quality problems have begun to occur. The ability to predict whether reservoirs may be more or less vulnerable to poor water quality and toxic cyanobacterial blooms, and why, is critical for reliable hazard prevention, planning and management. Therefore, our objectives were to develop an index of vulnerability to poor water quality based on simple measures of reservoir and catchment characteristics, and to test the index's ability to predict this vulnerability using water quality and cyanobacteria data collected from 15 drinking water reservoirs in subtropical southeast Queensland. We expected that an effective index would show positive correlation with nutrients, chlorophyll *a* concentrations and algal and cyanobacterial cell densities in summer months.

2 Materials and methods

2.1 Study region

The 15 reservoirs examined in this study supply drinking water to the urban and semi-rural populations of southeast Queensland (*c*. 2.8 million) and vary in catchment size and full supply capacity (Fig. 1, Table 1). In the past 30 years, average annual rainfall in the region has ranged from 800 to 1600 mm (www.bom.gov.au). Land use in the catchments is dominated by natural bushland and pastoral activities (mainly cattle grazing), with smaller proportions of cropping and residential lands (Fig. 2).

2.2 Index calculation

Reservoir and catchment characteristics can be summarised by several parameters, many of which have been examined for their ability to explain variation in reservoir water quality (e.g. Forbes et al. 2008). The parameters we used to calculate the vulnerability index (VI) satisfied the following four conditions: 1) correlation with water quality was well established in the literature, either theoretically or empirically; 2) parameters were easily calculated from readily available data on reservoir or catchments characteristics; 3) parameters were not strongly correlated with each other (Spearman correlation, $R < 0.70$ and/or $P > 0.05$), and 4) parameters were relatively static or predictable through time so that the index was unaffected by substantial spatial and temporal variation (e.g. this would exclude parameters like nutrient concentrations and water transparency). This last condition also ensured that the index was not self-forecasting (e.g. high nutrient concentrations predicting that a reservoir was vulnerable to high nutrient concentrations). Five parameters satisfied the conditions listed above and the VI, which ranges from 0 (lowest vulnerability) to 1 (highest vulnerability), was calculated as follows:

VI = (percentage grazing land cover^{a} + reservoir shoreline to surface area ratio^{bc} + reservoir volume at full supply capacity^{ab} + reservoir volume to catchment area ratio^{bc} + age since dam construction^{ab})/ 5

a) Range standardised so that the highest value = 1 and the lowest = 0;
b) Log transformed to reduce skew;
c) Range standardised so that the highest ratio = 0 and the lowest = 1.

Log transformation and range standardisation gave each parameter an equivalent weighting in the formula and created a comparative index of vulnerability among the 15 reservoirs.

2.3 Index testing

The ability of the index to predict the vulnerability of reservoirs to poor water quality and algal blooms was assessed by testing the correlation between index scores and water quality parameters in the 15 reservoirs. Each reservoir was sampled once between 9 February and 3 March 2009 in the late summer period. Over the past 30 years in the study area, mean rainfall and temperature during February has been 100-300 mm and 18-27 °C (min-max ranges, www.bom.gov.au). Heavy rainfall was experienced while sampling Kurwongbah reservoir and this rain event caused overflow at the dam walls of both Kurwongbah and Somerset at the time of sampling. There was also a bloom of the toxic cyanobacterium *Cylindrospermopsis raciborskii* in Borumba reservoir during sampling. Three reservoirs had destratification units in use near the dam wall to vertically mix the water: Leslie Harrison, Macdonald and North Pine (see Burford and O'Donohue 2006).

At least three sites were sampled in each reservoir. Sites were near the dam wall, mid-reservoir, and in the upstream section of each reservoir. Sampling was conducted at the deepest point of each site, which was at least 6 m (except for the upstream sites at Kurwongbah and Little Nerang which were 2-4 m in depth). Four sites were sampled in the larger reservoirs (e.g. Somerset) and in those with two major arms (Borumba, Hinze, Leslie Harrison and North Pine). In Wivenhoe, the largest reservoir out of the 15, only three sites were sampled due to an aquatic weed infestation restricting access to the most upstream reaches; however, the three sites were still representative of dam wall, mid-reservoir and upstream locations.

At each site, a 3 m depth-integrated sample of surface water was collected with a modified PVC pipe. Bottom water was collected with a van Dorn (3.2 L Vertical Beta Plus) sampler from 1 m above the bottom of each site. Surface and bottom water were each transferred to a clean bucket from which samples for water quality analyses were taken. Each process was repeated to obtain duplicate samples.

Surface water was subsampled for analyses of total nitrogen and phosphorus (TN, TP; for both whole and dissolved fractions), dissolved inorganic nitrogen and phosphorus (DIN and DIP), and chlorophyll *a* (Chl) concentrations as well as algal identification and cell densities (cells mL^{-1}), including potentially toxic cyanobacterial species (*Anabaena* *circinalis, A. bergii, Aphanizomenon ovalisporum, C. raciborskii* and *Microcystis aeruginosa*). Bottom water subsamples were analysed for these same parameters, excluding chlorophyll *a* concentration and algal identification. All subsamples that were analysed for dissolved fractions were pre-filtered, *in situ*, through 0.45-μm membrane filters (Millipore) and subsamples for algal identification were fixed with Lugol's iodine solution (0.6 % final concentration). Concentrations of dissolved organic nutrients (DON and DOP) were determined by subtraction (e.g. DON = dissolved TN fraction – DIN). All samples were stored in the dark and on ice until transported to the laboratory.

Nutrient analyses were conducted following standard colorimetric methods; chlorophyll *a* was extracted in 100% acetone and measured spectrophotometrically (American Public Health Association, 1995). DIN and DIP concentrations were often near or at detection limit (0.002 mg L^{-1}) and were not used in further analyses. Algal taxa were identified to species level, where possible, under 400x phase-contrast microscopy and cells were counted using a Sedgewick Rafter counting chamber (Lund et al. 1958; Burford et al. 2007).

The top layer (*c*. 2 cm) of sediments at the bottom of each site was collected in duplicate using a weighted sediment corer. These samples were analysed for TN and TP concentrations as well as stable carbon and nitrogen isotope ratios ($\delta^{13}C$ and $\delta^{15}N$ ‰), which were determined using a mass spectrometer (GV Isoprime, Manchester, UK), following standard methods (American Public Health Association 1995; Jardine et al. 2003).

Data was subjected to Spearman's rank correlation as this test handles parameters with skewed distributions and/or heteroskedacity, which included the majority of the water quality parameters used in the analysis as well as the VI itself. Statistical significance was recorded at $P < 0.05$ and tests were conducted within the R stats package (www.r-project.org/). The strength (size of the correlation coefficient, R) and significance of correlations between VI and water quality parameters were also compared with those between the individual VI (grazing, shore:SA, Vol, Vol:CA, age) and water quality parameters.

Data from a previous study on 7 of the 15 reservoirs were also used as an independent dataset to assess the VI's effectiveness (Burford et al. 2007). In this study, water quality was assessed at two sites (dam wall and upstream) in each reservoir during spring (October 2004), early summer (December 2004), late summer (February 2005) and autumn (April 2005). Correlations between these data and the VI were tested using the same methods outlined for the February 2009 study, with the VI calculated based on reservoir age in 2004 and 2005 for the corresponding years of sampling.

3 Results

3.1 Water quality and algal cell densities (Feb 2009)

Mean surface water temperatures, integrated over the top 3 m, ranged from 26.7 ± 0.5 °C at Baroon reservoir to 28.7 ± 0.7 °C at Moogerah. The exception was Little Nerang reservoir, which had a mean surface water temperature of 22.5 ± 1.8 °C. Within each reservoir, TN and TP concentrations were generally higher in bottom waters than surface waters (see Accessory Publication). Borumba, Moogerah and Manchester had the highest mean concentrations of TN, in both surface (0.66-0.88 mg L^{-1}) and bottom waters (1.18-1.31 mg L^{-1}). TP concentrations were similar among most reservoirs; mean concentrations were all < 0.05 mg L^{-1} except for the bottom waters of Borumba, Somerset, Wivenhoe, Maroon and Moogerah (0.135-0.372 mg L^{-1}). Many reservoirs had high proportions of DON in surface waters and high mean concentrations of DOP were found in bottom waters of Borumba, Wivenhoe, Somerset, Maroon and Moogerah (0.094-0.271 mg L^{-1}).

The algal composition of surface water samples from all reservoirs, in terms of mean cell densities, was dominated by cyanobacteria (Fig. 3; Accessory Publication). Borumba had the highest mean algal cell density of all reservoirs (455 000 ± 91 000 cells mL^{-1}), followed by Moogerah, Somerset and Wivenhoe (means > 200 000 cells mL^{-1}). Hinze had the lowest mean algal cell density (20 000 ± 5 000 cells mL^{-1}). Potentially toxic cyanobacteria, when present within reservoirs, were most often dominated by *C. raciborskii*. The exceptions were Hinze (mid-reservoir only) and Maroon, in which *Anabaena circinalis* dominated. Five reservoirs had no potentially toxic cyanobacterial species identified (Cooloolabin, Macdonald, Ewen Maddock, Leslie Harrison and Little Nerang).

3.2 Index of vulnerability (VI)

Based on the index calculation outlined in the Methods, Wivenhoe reservoir had the highest level of vulnerability to poor water quality and algal blooms (VI = 0.77), followed closely by Somerset (0.74) and Moogerah (0.67) (Fig. 4). Cooloolabin had the lowest VI (0.18). Ewen Maddock (0.29), Little Nerang (0.30), Hinze (0.33) and Leslie Harrison (0.36) also had low indices. The remaining reservoirs had intermediate VI scores (Macdonald = 0.43, Manchester = 0.44, Baroon = 0.46, Maroon = 0.48, Kurwongbah and North Pine = 0.50, Borumbah = 0.52).

3.3 Index performance

Statistically significant correlations between water quality parameters and the VI scores were all positive (Table 2). These correlations were also stronger and more often statistically significant than the correlations between water quality parameters and each of the five parameters used to calculate the VI (Table 3). Significant correlations were found between the VI and both the densities and proportions of algal cells in all study periods, except October 2004 (early spring) for which correlations were all non-significant. Among the significant correlations, the strongest were with total algal and cyanobacterial cell densities, and the strongest of these were found in February 2009 (R = 0.82 and 0.83) and December 2004 (R = 0.86 and 0.86). Correlations with the proportion of cyanobacterial cells were strongest during the 2004-5 study (R = 0.74-0.82). February was the only month for which significant correlations with potentially toxic species were detected (2005: R = 0.69 for density only; 2009: R = 0.71 and 0.64 for density and proportion respectively). This was also the case for chlorophyll *a* concentrations (February 2005, R = 0.56; February 2009, R = 0.41). In addition, correlations with nutrient concentrations measured in all months (water column TN and TP) were only significant in February 2009 (except for surface TN in December 2004; R = 0.55) and the strongest was with TP in bottom waters (R = 0.82). The VI also correlated well with sediment nutrient and carbon data, in particular with $\delta^{13}C$ (R = 0.72), which were measured in February 2009 only.

4 Discussion

The index of vulnerability to poor water quality and algal blooms correlated strongly and significantly with algal cell densities, including potentially toxic cyanobacteria, and the proportions of cyanobacteria within the subtropical reservoirs during summer months. Cyanobacteria are capable of regulating their buoyancy, surviving low light conditions, storing nutrients and utilising forms of nutrient that are inaccessible to other taxa, all of which allows them to dominate the algal community under various physicochemical conditions (Padisák 1997; Burford et al. 2006; Posselt et al. 2009). Given this flexibility, the ability of the VI to reflect increased summer densities and proportions of cyanobacteria, based on physical characteristics of the reservoirs and catchments alone, suggests that it may be more capable of assessing bloom susceptibility than traditional measures based on nutrient concentrations, trophic status or light availability (e.g. Downing et al. 2001). Our index also performed better than each of the VI parameters alone, which further supported its ability to comparatively assess the reservoirs' vulnerability. Overall, our analyses suggest that strong links exist among the physical environment of dammed river systems, their physicochemical characteristics and algal ecology, although further work is required to understand and show causality.

Land use, particularly animal agriculture, has been implicated in the eutrophication of streams, rivers, lakes and reservoirs the world over (Søndergaard and Jeppesen 2007). In our study of subtropical reservoirs, 12 out of the 18 water quality parameters analysed showed significant correlation with the percentage of grazing land cover in the reservoirs' catchments. Reservoirs with lower reservoir volume to catchment area ratios are more likely to have stronger links with catchment characteristics, including land use and the consequent reduction in water quality, than those with higher ratios, regardless of climatic zone (cf. Burford et al. 2007). Indeed, lower ratios have been linked with higher concentrations of chlorophyll *a* and total phosphorus in temperate-zone reservoirs, Ohio USA (Knoll et al. 2003).

Physical characteristics of reservoirs also affect internal water quality. For example, increased water residence time, through the increased net loading of nutrients in reservoirs, has long been implicated in the promotion of algal blooms and reduced water quality (Søballe and Kimmel 1987; Harris 2001). We did not include this parameter in our index, however, as the data needed to calculate residence time was not available for all reservoirs. In addition, water levels in the reservoirs fluctuate through time due to variation in inflow and outflow volumes and timing, such that residence time is not constant. Rather, we used age since completion of the dam wall as an alternative indicator of the nutrient loading capacity of reservoirs. Older reservoirs were assumed to have increased stores of nutrients, and strong correlations between nutrient concentrations in the water column and reservoir age were found. The specific processes linking reservoir age to present-time water quality are not clear; however, it may be that as reservoirs age, sediment loading into reservoirs results in siltation and reduced water depth, particularly in the upper reaches. Nutrients released from sediments would therefore be more readily available for algal growth in surface waters, consistent with increased benthic-pelagic coupling (see also Nõges et al. 1999).

In addition, our index was based on the ratio of reservoir shoreline length to surface area. Reservoirs with lower ratios (shoreline length to surface area or reservoir volume) are likely to have a stronger pelagic and hypolimnic influence on reservoir water quality and ecosystem processes than those with higher ratios (Wetzel 2001). For the tropics and subtropics in particular, reservoirs that have a greater proportion of pelagic than littoral habitat may become more susceptible to poor water quality when internal processes, such as stratification and sediment remineralisation, start to affect water quality in the summer months (Jones and Poplawski 1998).

The combination of these parameters (percentage grazing, shoreline to surface area ratio, reservoir volume, volume to catchment area ratio and reservoir age) produced a good index of vulnerability to poor water quality and algal blooms in the subtropical reservoirs, and in particular, to increased cyanobacterial densities and proportions in summer months. Correlation between the VI and pre-summer water quality (Oct 2004) was not detected. Algal composition during this month was significantly different to summer and post-summer months (Burford and Donohue 2006; Burford et al. 2007) and recent studies of Wivenhoe reservoir suggest that this pre-summer/summer difference may be expected for other reservoirs in the study region (P. Muhid, unpublished results). As expected, correlations were highly significant in summer months (Dec 2004, Feb 2005, Feb 2009), which included both small (n= 14) and larger (n = 42-50) datasets. The index also showed positive correlation with nutrient concentrations and/or stable isotope ratios in the water column and sediments measured during the 2009 summer sampling period. Warm temperatures and stratification can lead to sediment remineralisation and the renewed availability of nutrients to cyanobacteria. In addition, enriched carbon and nitrogen isotope ratios in reservoirs have been linked to increased autotrophic production (for carbon) and inputs of nitrogen associated with agricultural or urban land use (Leavitt et al. 2006; Wu et al. 2006; Tomaszek et al. 2009).

In summary, our analysis indicated that the VI may be useful for assessing summer bloom vulnerability in subtropical reservoirs. However, the link between summer rainfall and reservoir water quality, like that with summer temperatures, was inherent in the reasoning behind the index. Southeast Queensland experienced drought conditions (minimum summer rainfall) between *c*. 2002 and December 2009. In conjunction with summer temperatures, the recent rainfall and inflow events prior to the February 2009 sampling period, although not substantial, may have improved the VI's performance in comparison with the other sampling periods, including February 2005. As such, the risk of summer bloom events in reservoirs with high index scores may decrease in drought conditions. However, if water depth declines to a critical threshold where sediment remineralisation processes promote algal growth throughout the water column, summer blooms may be inevitable (P. Muhid, unpublished results), particularly given the known effect of increased temperatures on algal growth.

The ultimate aim of the VI is to provide water authorities and managers with a rapid tool to confidently assess how vulnerable a reservoir is (or may be) to poor water quality, and in particular, cyanobacterial blooms. For example, we applied the index to a new dam (Wyaralong) being constructed about 20 km northeast of Maroon reservoir, to compare its potential vulnerability with the 15 reservoirs examined above (Fig. 5). Construction is scheduled for completion by end 2011. Based on planned dimensions and current grazing cover (46.9%), Wyaralong's VI (0.55) predicts mid-range vulnerability, comparative with reservoirs like Maroon and Kurwongbah, for at least 5 years after competition (Table 1, Fig. 5). However, 10 years after completion, the VI is more comparable with that of Baroon, and 20 years after completion with Borumbah and North Pine. After 100 years, the VI is at the higher end of vulnerability to eutrophication and cyanobacterial blooms, such that Wyaralong is the fourth most vulnerable reservoir with respect to Moogerah, Somerset and Wivenhoe (Fig. 5).

A simple exercise in decreasing or increasing the percentage of grazing land cover in Wyaralong's catchment by 10% via reforestation, predicts that the VI will either reach the same endpoint (the fourth most vulnerable reservoir) after only 5 years (+ 10% grazing cover) or remain below this point for at least 100 years (- 10% grazing cover) (all other parameters except age were unchanged for all reservoirs; Fig. 5). This demonstrates that the VI could provide input to the planning of new reservoirs and assist in decision making about investment to mitigate for adverse water quality outcomes. This may include such comparisons as costs of land use change versus increased treatment and may lead to the expansion of impact assessments to include the possibility of new reservoirs meeting water quality targets and to consider the potential impacts of algal blooms.

Our paper encapsulates the physical characteristics of a group of reservoirs and their catchments into an effective indicator of the potential for summer blooms and water quality issues. However, the ability of the VI to predict the comparative susceptibility to summer blooms of cyanobacteria and eutrophic conditions was assessed for a limited number of reservoirs and in the subtropics alone. Adaptations may be required to achieve an acceptable level of correlation between the VI and water quality parameters in any one set of reservoirs (e.g. using residence time instead of reservoir age to calculate the index). To confirm the generic usefulness of the VI, similar tests are recommended in other reservoirs within subtropical, tropical and even temperate climates.

5 Conclusions

- This is the first index to encapsulate the physical characteristics of subtropical reservoirs and their catchments into an effective indicator of summer bloom vulnerability.

- The index of vulnerability to poor water quality and cyanobacterial blooms in the subtropical reservoirs examined in this study was based on the percentage of agricultural land use in catchments and physical characteristics of reservoirs.
- The index correlated strongly with increased cyanobacterial cell densities in summer months, as well as their proportional contribution to the total algal density.
- The index has the capability to predict vulnerability to poor water quality and summer blooms of cyanobacteria in subtropical, and potentially, tropical and temperate-zone reservoirs. With climate change, continued river impoundment and the growing demand for potable water, our index may provide decision making support when planning reservoirs, in the subtropics and elsewhere, to reduce their vulnerability to cyanobacterial bloom events.

Acknowledgements

We thank rangers, boat drivers and water quality monitoring staff at Seqwater; Stephen Faggotter and Deb Gale for their assistance with sampling; Queensland Forensic and Scientific Services for nutrient and chlorophyll analyses and algal identification, in particular Gary Prove, Eugene Lee, Priya Muhid and Karen Reardon; Jason Kerr and Rene Diocares for stable isotope sample preparation and analysis at Griffith University; Seqcatchments (Brisbane), Monica Poel and Leanne Bowen for catchment boundary and land use spatial data, which was based on Statewide Landcover and Trees Study data from the State of Queensland (Department of Natural Resources and Water) 2006 (http://www.nrw.qld.gov.au/slats/index.html). This work was supported and funded by Queensland Bulk Water Supply Authority (Seqwater). We are grateful to two anonymous reviewers for their comments on our manuscript.

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Figure captions

Figure 1: Dam wall locations of 15 reservoirs (closed triangles) in subtropical southeast Queensland, Australia, sampled during late summer 2009, and a new reservoir currently in construction (open triangle), shown with catchment boundaries of the major river systems in which the reservoirs are located.

Figure 2: Proportions of different land use cover in the catchments of 15 reservoirs. See Table 1 for the key to reservoir coding.

Figure 3: Algal densities (cells mL^{-1}) within reservoirs (means with standard errors as bars) sampled in February 2009. Potentially toxic cyanobacteria include *Anabaena circinalis*, *Aphanizomenon ovalisporum*, *Cylindrospermopsis raciborskii* and *Microcystis aeruginosa*.

Figure 4: Vulnerability Index for 15 reservoirs in subtropical Queensland (based on ages of reservoirs in 2009). See Table 1 for the key to reservoir coding.

Figure 5: Vulnerability Index for 16 reservoirs in subtropical Queensland, at one, five, twenty and one hundred years since the planned completion of Wyaralong dam wall in 2011, given: the current percentage of grazing land cover in Wyaralong catchment (top row); minus 10% (middle row); plus 10% (bottom row). Unbroken arrows show the change in the level of among-reservoir vulnerability between grazing cover scenarios. Broken arrows show the change in among-reservoir vulnerability through time. See Table 1 for the key to reservoir coding.

Table 1: Physical characteristics of 16 subtropical reservoirs and their catchments.

Code	Reservoir	Shore (km)	SA (km^2)	Vol (ML)	Shore:SA (km^{-1})	Shore:Vol (km-2)	Depth (m)	Age (y)	CA (ha)	Vol:CA (ML ha^{-1})
Bar	Baroon	25.5	3.9	61000	6.6	4.2	15.7	21	6530	9.3
Bor	Borumba	44.5	3.8	46000	11.9	9.7	12.2	46	46900	1.0
Cool	Cooloolabin	12.8	1.4	14200	8.9	9.0	9.9	30	736	19.3
Ewen	Ewen Maddock	14.3	2.3	16600	6.4	8.6	7.4	27	2130	7.8
Hin	Hinze	67.4	9.3	163000	7.2	4.1	17.5	20	17600	9.3
Kur	Kurwongbah	33.4	3.4	14400	9.9	23.3	4.2	40	5250	2.7
LHD	Leslie Harrison	34.5	4.2	24800	8.2	13.9	5.9	25	8890	2.8
LNer	Little Nerang	10.9	0.6	9280	19.6	11.8	16.6	48	3600	2.6
Mac	Macdonald	38.3	2.6	8000	14.7	47.9	3.1	29	4960	1.6
Man	Manchester	27.3	2.6	26000	10.5	10.5	10.0	93	7260	3.6
Mar	Maroon	17.9	3.3	44300	5.4	4.0	13.3	35	10500	4.2
Moog	Moogerah	29.3	7.7	83800	3.8	3.5	10.9	48	22700	3.7
NPD	North Pine	167	21.2	215000	7.9	7.8	10.1	33	34900	6.2
Som	Somerset	237	39.7	369000	6.0	6.4	9.3	56	133000	2.8
Wiv	Wivenhoe	462	110	1150000	4.2	4.0	10.5	25	568000	2.0
Wya	Wyaralong	110	1.1	103000	9.0	10.8	8.4	-	54590	1.9

Shore, shoreline perimeter; SA, reservoir surface area at full supply level (FSL); Vol, water storage capacity at FSL; Depth (mean) = Vol/SA; Age, age to 2009 since the completion of dam construction; CA, catchment area; -, Wyaralong dam due for completion in 2011.

Table 2: Correlation between the VI and water quality parameters (mg L^{-1} unless otherwise indicated) within 15 subtropical reservoirs. Bold text indicates significant correlations ($P < 0.05$).

		Oct 2004*			Dec 2004*			Feb 2005*			Apr 2005*			Feb 2009		
		R	*P*	n	R	*P*	n	R	*P*	n	R	*P*	n	R	*P*	n
Algae (cells mL^{-1})	S	0.14	0.6727	14	**0.86**	0.0001	14	**0.73**	0.0027	14	**0.61**	0.0010	14	**0.82**	<0.0001	50
Cyano (cells mL^{-1})	S	0.10	0.7633	14	**0.86**	0.0001	14	**0.75**	0.0019	14	**0.59**	0.0013	14	**0.83**	<0.0001	50
Cyano (%)	S	0.10	0.7175	14	**0.74**	0.0023	14	**0.76**	0.0015	14	**0.68**	0.0003	14	**0.37**	0.0064	50
Toxic (cells mL^{-1})	S	0.20	0.5014	14	0.20	0.4797	14	**0.69**	0.0061	14	0.00	0.9400	14	**0.71**	<0.0001	50
Toxic (%)	S	0.35	0.2293	14	0.00	0.9758	14	0.44	0.1205	14	0.03	0.5543	14	**0.64**	<0.0001	50
Chl (μg L^{-1})	S	0.17	0.5621	14	0.28	0.3260	14	**0.56**	0.0394	14	0.03	0.5634	14	**0.41**	0.0028	50
TN	S	0.51	0.0652	14	**0.55**	0.0414	14	0.50	0.0676	14	0.22	0.0933	14	**0.66**	<0.0001	50
TN	B	0.33	0.2520	14	0.40	0.1574	14	0.10	0.7743	14	0.03	0.5861	14	**0.50**	0.0002	49
TN (mg kg^{-1})	Sed			0			0			0			0	0.10	0.4905	48
TP	S	0.30	0.3017	14	0.10	0.7515	14	0.10	0.7978	14	0.01	0.7504	14	**0.50**	0.0002	50
TP	B	0.17	0.5856	14	0.33	0.2386	14	0.26	0.3747	14	0.13	0.2018	14	**0.82**	<0.0001	49
TP (mg kg^{-1})	Sed			0			0			0			0	**0.51**	0.0002	48
DON	S			0			0			0			0	**0.55**	<0.0001	50
DON	B			0			0			0			0	**0.28**	0.0454	49
DOP	S			0			0			0			0	0.10	0.4886	50
DOP	B			0			0			0			0	**0.66**	<0.0001	49
$\delta^{13}C$ (‰)	Sed			0			0			0			0	**0.72**	<0.0001	42
$\delta^{15}N$ (‰)	Sed			0			0			0			0	**0.57**	<0.0001	42

* Raw data sourced from a previous study (Burford et al. 2007). Cyano, cyanobacteria; Toxic, potentially toxic cyanobacteria; S, surface; B, bottom; Sed, sediment.

Table 3: A comparison of correlation between the VI versus individual physical parameters and water quality within 15 subtropical reservoirs sampled in February 2009. Bold text indicates significant correlations ($P < 0.05$).

			Grazing		Shore:SA		Vol		Vol:CA		Age		VI	
		n	R	*P*	R	*P*	R	*P*	R	*P*	R	*P*	R	*P*
Algae (cells mL^{-1})	S	50	**0.41**	0.0029	-0.26	0.0698	**0.53**	0.0001	**-0.47**	0.0006	**0.51**	0.0001	**0.83**	0.0000
Cyano (cells mL^{-1})	S	50	**0.47**	0.0006	**-0.35**	0.0115	**0.56**	0.0000	**-0.41**	0.0028	**0.45**	0.0009	**0.83**	0.0000
Cyano (%)	S	50	**0.47**	0.0007	**-0.65**	0.0000	**0.36**	0.0108	0.10	0.4764	0.04	0.8083	**0.38**	0.0064
Toxic (cells mL^{-1})	S	50	**0.34**	0.0163	-0.06	0.6921	**0.50**	0.0002	-0.26	0.0733	**0.35**	0.0126	**0.70**	0.0000
Toxic (%)	S	50	**0.32**	0.0255	0.00	0.9992	**0.44**	0.0015	-0.22	0.1319	**0.29**	0.0389	**0.64**	0.0000
Chl ($\mu g\ L^{-1}$)	S	50	0.14	0.3237	**0.36**	0.0095	0.05	0.7532	**-0.62**	0.0000	**0.57**	0.0000	**0.41**	0.0028
TN ($mg\ L^{-1}$)	S	50	0.17	0.2276	-0.05	0.7136	0.22	0.1023	**-0.48**	0.0005	**0.53**	0.0001	**0.66**	0.0000
TN ($mg\ L^{-1}$)	B	49	-0.04	0.7829	-0.18	0.2004	0.23	0.1265	-0.19	0.1779	**0.67**	0.0000	**0.50**	0.0002
TN ($mg\ kg^{-1}$)	Sed	48	-0.07	0.6491	0.11	0.2628	**0.48**	0.0004	-0.03	0.8373	0.25	0.0854	-0.10	0.4905
TP ($mg\ L^{-1}$)	S	50	**0.46**	0.0008	0.04	0.7591	0.17	0.2404	**-0.50**	0.0002	**0.33**	0.0185	**0.50**	0.0002
TP ($mg\ L^{-1}$)	B	49	**0.59**	0.0000	**-0.42**	0.0025	**0.53**	0.0001	**-0.39**	0.0050	**0.38**	0.0064	**0.82**	0.0000
TP ($mg\ kg^{-1}$)	Sed	48	**0.42**	0.0026	**-0.47**	0.0000	**0.48**	0.0004	-0.03	0.8373	0.08	0.5665	**0.51**	0.0002
DON ($mg\ L^{-1}$)	S	50	0.05	0.7519	-0.09	0.5319	0.14	0.3189	**-0.35**	0.0117	**0.50**	0.0002	**0.55**	0.0000
DON ($mg\ L^{-1}$)	B	49	0.16	0.2744	-0.02	0.8914	0.04	0.7948	-0.24	0.0914	0.19	0.6229	**0.28**	0.0454
DOP ($mg\ L^{-1}$)	S	50	**0.28**	0.0456	-0.26	0.0677	0.03	0.8251	-0.15	0.2946	0.07	0.1820	0.10	0.4886
DOP ($mg\ L^{-1}$)	B	49	**0.47**	0.0005	**-0.39**	0.0056	**0.50**	0.0002	**-0.29**	0.0379	**0.31**	0.0260	**0.66**	0.0000
$\delta^{13}C$ (‰)	Sed	42	**0.74**	0.0000	**-0.28**	0.0043	**0.44**	0.0015	-0.12	0.3963	0.10	0.5050	**0.72**	0.0000
$\delta^{15}N$ (‰)	Sed	42	**0.62**	0.0000	**-0.38**	0.0061	**0.71**	0.0000	0.10	0.4861	-0.22	0.1282	**0.58**	0.0000

Cyano, cyanobacteria; Toxic, potentially toxic cyanobacteria; S, surface; B, bottom; Sed, sediment.

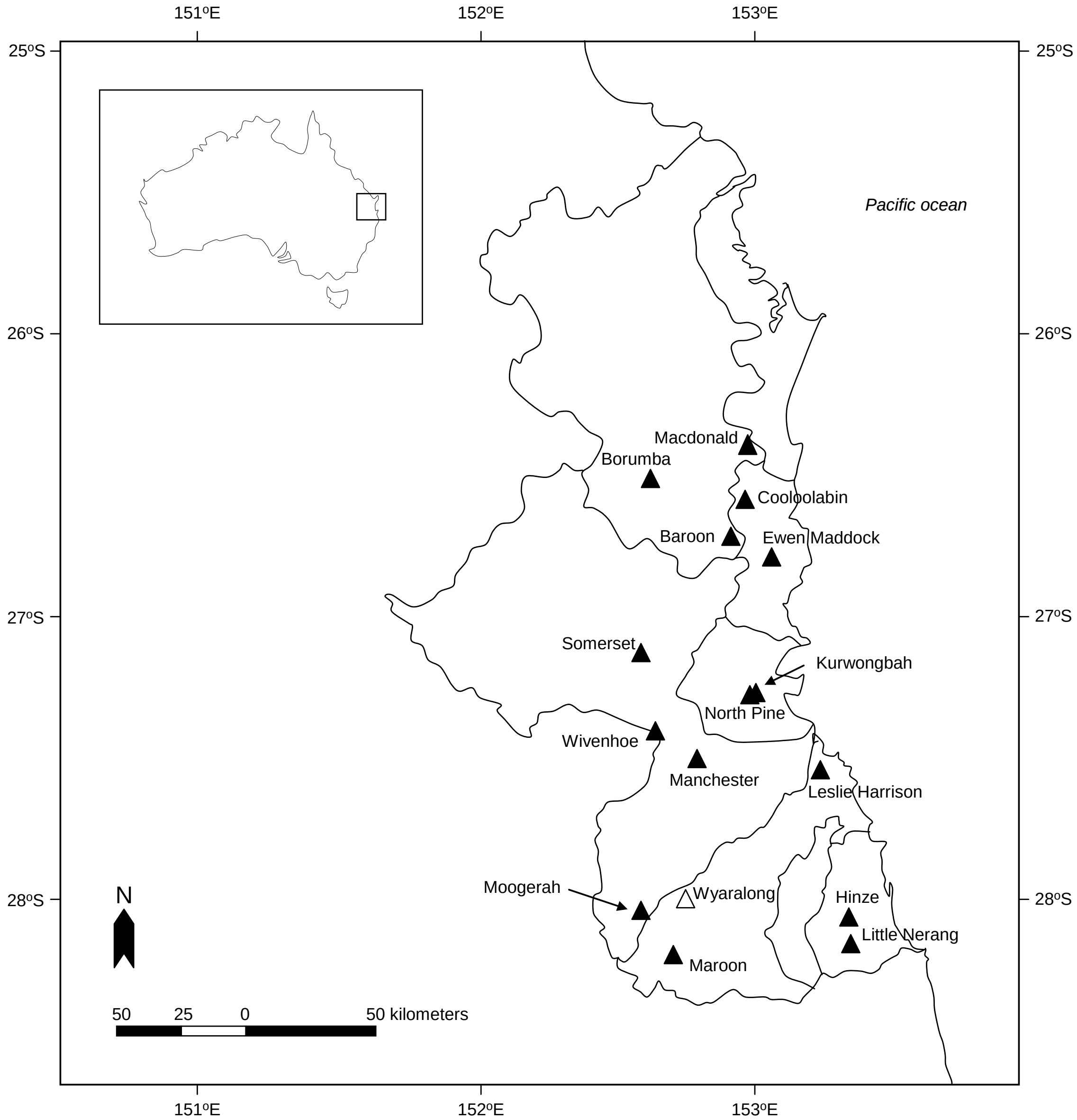

151ºE
152ºE
153ºE
25ºS
26ºS
27ºS
28ºS
Pacific ocean
Macdonald
Borumba
Cooloolabin
Baroon
Ewen Maddock
Somerset
Kurwongbah
North Pine
Wivenhoe
Manchester
Leslie Harrison
Moogerah
Wyaralong
Hinze
Little Nerang
Maroon
N
50
25
0
50 kilometers

Figure 1

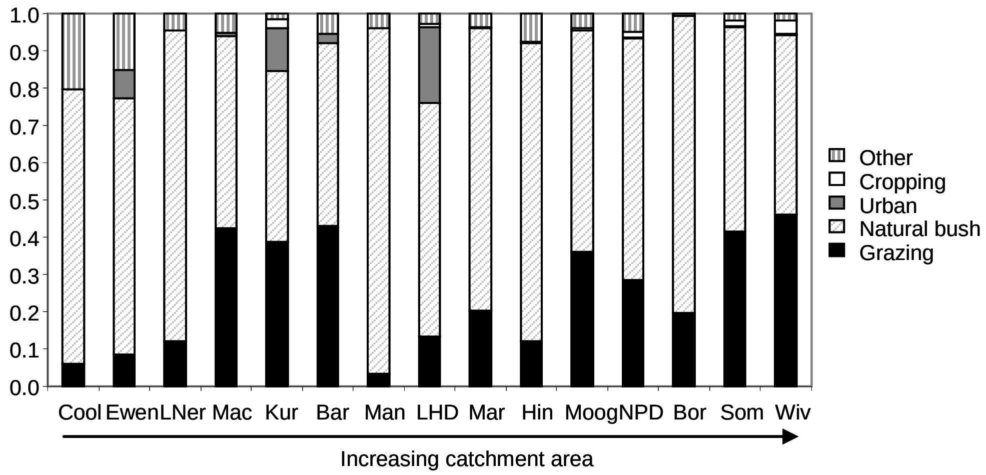

1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
Cool
Ewen
LNer
Mac
Kur
Bar
Man
LHD
Mar
Hin
Moog
NPD
Bor
Som
Wiv
Increasing catchment area
Other
Cropping
Urban
Natural bush
Grazing

Figure 2

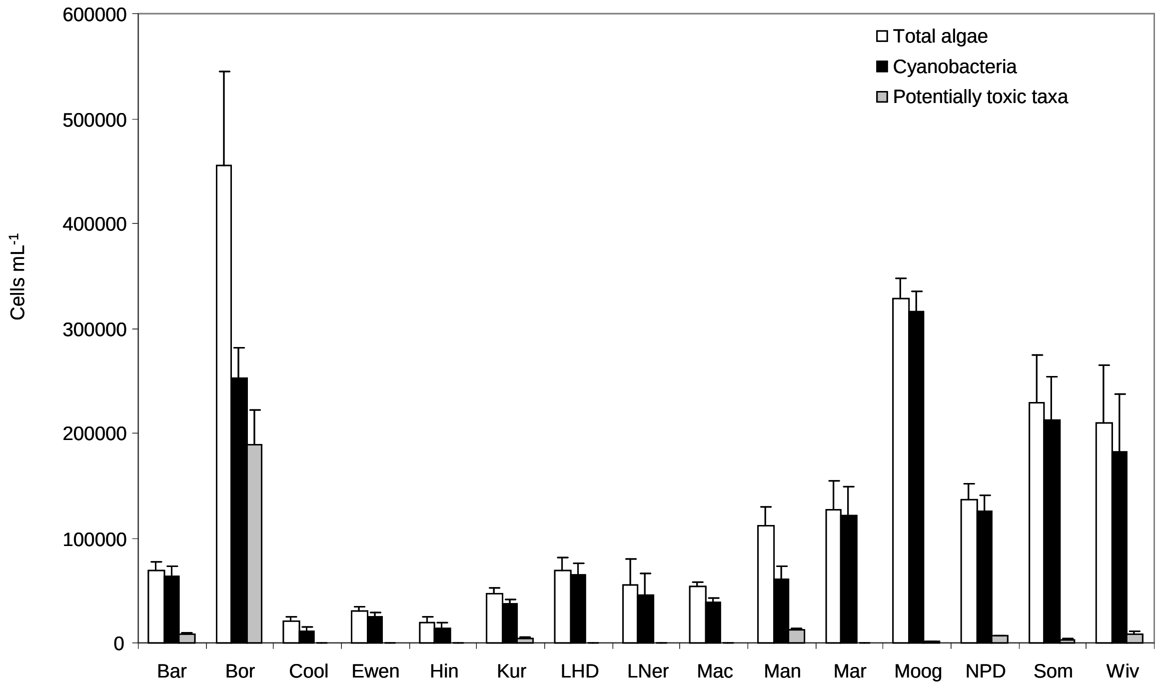
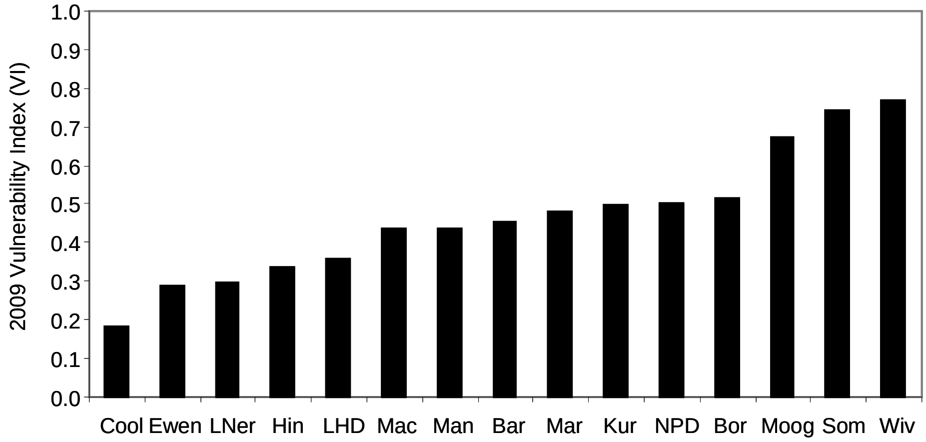
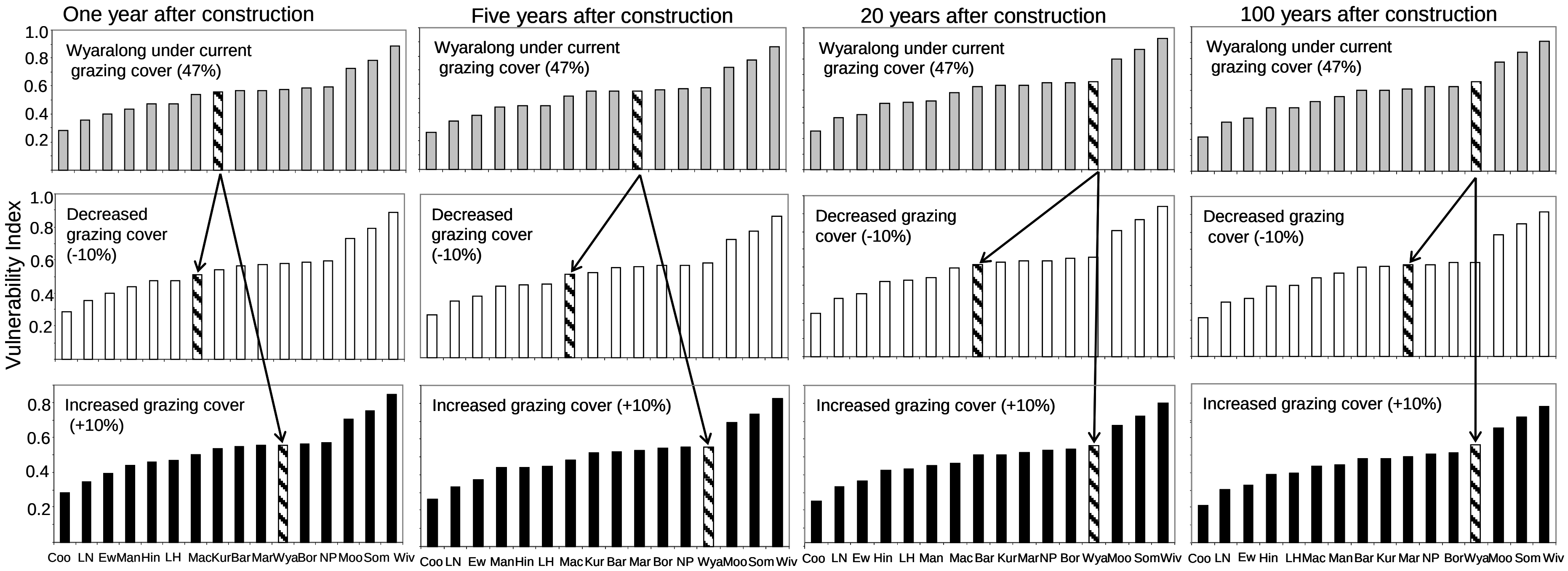

600000
500000
400000
300000
200000
100000
0
Cells mL-1
Total algae
Cyanobacteria
Potentially toxic taxa
Bar
Bor
Cool
Ewen
Hin
Kur
LHD
LNer
Mac
Man
Mar
Moog
NPD
Som
Wiv

Figure 3



2009 Vulnerability Index (VI)
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
Cool Ewen LNer Hin LHD Mac Man Bar Mar Kur NPD Bor Moog Som Wiv

Figure 4



One year after construction
Five years after construction
20 years after construction
100 years after construction
Vulnerability Index
Wyaralong under current grazing cover (47%)
Decreased grazing cover (-10%)
Increased grazing cover (+10%)
1.0
0.8
0.6
0.4
0.2
Coo LN Ew Man Hin LH Mac Kur Bar Mar Wya Bor NP Moo Som Wiv
Coo LN Ew Man Hin LH Mac Kur Bar Mar Bor NP Wya Moo Som Wiv
Coo LN Ew Hin LH Man Mac Bar Kur Mar NP Bor Wya Moo Som Wiv
Coo LN Ew Hin LH Mac Man Bar Kur Mar NP Bor Wya Moo Som Wiv

Figure 5