Deriving surface water quality closure criteria for natural waterbodies adjacent to an Australian uranium mine

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Abstract

Georgetown Billabong (GTB), a natural waterbody located adjacent to the Ranger Uranium Mine (NT), has received low levels of minesite solutes since the start of mining. Surface water quality closure criteria for GTB are being derived from those periods of the water quality record that support an ecological condition (viz macroinvertebrate community data) equivalent to that found in regional reference billabongs, an approach consistent with the ANZECC/ARMCANZ (2000) water quality guidelines. GTB macroinvertebrates sampled in 1995, 1996 and 2006 were similar to reference, resulting in derived water quality criteria protective of biota yet not as conservative had these been derived using a reference condition approach. GTB water quality deteriorated in 2010–11 and macroinvertebrates sampled in this period were no longer similar to reference; corresponding water quality data, therefore, could not be used to update closure criteria.

Key words: uranium mining, closure criteria, water quality guidelines, ecotoxicology, biological indicators, Australia

Introduction

Georgetown Billabong (GTB) is a natural waterbody located immediately downstream of the Ranger Uranium Mine, in northern Australia, and discharges into Magela Creek. It has historically received low levels of minesite solutes (mainly magnesium sulfate and also uranium,) since the inception of mining in the early 1980s. GTB has a long water quality record and is being used as a case study to develop surface water quality closure criteria for natural waterbodies within the mining lease. Figure 1 shows the historical seasonal electrical conductivity record (largely reflecting the concentration of magnesium sulfate). Decommissioning of the minesite is currently scheduled to start in 2020.

The surface water quality closure criteria will inform the design of the final landform in the context of the engineering works needed to limit the total load of solutes (via surface and ground-water pathways) and suspended sediments leaving the site. The setting of surface water quality criteria will set a limit on allowable additions of solutes arising from such features as mine pits that have been backfilled with tailings. Surface water criteria are a thus key requirement for the closure planning process, and need to be specified as early as possible so that the required engineering works can be designed and financially provisioned for (Jones et al. 2008).

The Australian and New Zealand Water Quality Guidelines (ANZECC & ARMCANZ 2000) approach to deriving water quality criteria, from local biological response data, provides the model for surface water closure criteria derivation.

Environmental requirements for Ranger specify that the ecological condition of GTB and other waterbodies near the minesite, post-closure, is to be consistent with similar undisturbed (reference) billabongs in the adjacent, high-conservation-value, Kakadu National Park (KNP). With this stipulation, then the range of measured water quality data from GTB over time that supports such an ecological condition (as measured by macroinvertebrate communities in this instance) may be used to derive the closure criteria. GTB has received inputs of mine-derived solutes over time and could thus be categorised as a mine-disturbed waterbody (albeit to a low level). However, there are periods in its recent history when, despite mine-affected water quality, the biological condition of GTB is similar to reference condition. Water quality closure objectives derived for the waterbody, therefore, will not be as conservative as those resulting from an alternative approach of conformance with data from a reference waterbody or historical reference condition. Nevertheless, and consistent with the ANZECC/ARMCANZ framework, they will maintain a sufficiently conservative level of protection for the resident aquatic biota provided that the biological condition of the (slightly) disturbed waterbody is indistinguishable from reference (unimpacted) waterbodies.

Macroinvertebrate sampling conducted in the littoral macrophyte (water column) habitat in 1995, 1996 and 2006 supported the conclusion that biological conditions (macroinvertebrate families and their relative abundances) in GTB conformed to those of reference waterbodies sampled elsewhere in the region. These results indicated that the historical water quality regime in GTB was compatible with the maintenance of the aquatic ecosystem values of KNP. Derived water quality closure criteria for electrical conductivity (a surrogate for the main mine-derived solute, $MgSO_4$), magnesium and uranium were reported in Jones et al. (2008), based on water quality and aquatic macroinvertebrate data acquired until 2006.

Since 2006 when the last macroinvertebrate sampling of regional billabongs occurred, the primary trigger for further assessment of biological 'health' of these waterbodies has been the deterioration in water quality in GTB beyond the quality observed in past sampling years (1995, 1996 and 2006). This occurrence provides an opportunity to adjust the previously-derived closure criteria in the context of an adaptive framework for criterion derivation.

In became evident in late 2010 that the late dry season water quality in GTB (viz electrical conductivity) had worsened beyond the quality observed in the past decade (Figure 1). This prompted further biological sampling and accordingly, macroinvertebrate communities from thirteen waterbodies (same sites as in 2006) were re-sampled during the late wet /early dry season period in 2011.

Methods

For the period from 1995 to 2011, macroinvertebrate communities have been sampled from up to 14 lentic waterbodies after the summer wet season (typically in the wet-dry transition period, April and May). The waterbodies are located in two catchments, Magela and Nourlangie Creeks, and include:

- *Mine-water-exposed waterbodies*: The natural Djalkmara, Coonjimba, Georgetown and Gulungul Billabongs and the minewater management ponds, Ranger Retention Pond 1 (RP1) and Retention Pond 2 (RP2);
- *Reference sites, not exposed to Ranger mine waters*: Jabiru Lake and Baralil, Corndorl and Wirnmuyurr Billabongs (Magela Creek catchment); Malabanjbanjdju, Anbangbang, Buba and Sandy Billabongs (Nourlangie Creek catchment).

After 1996, Djalkmara Billabong ceased to exist as a natural waterbody following commencement of mining of Ranger orebody 3. The locations of the waterbodies referred to in this paper are available on the Supervising Scientist Division's website (see Maps 2 and 3 of http://www.environment.gov.au/ssd/publications /ss10-11/pubs/annual-report-10-11.pdf).

In each waterbody, samples were collected from five sites and at each site, separate samples were taken from the surface waters amongst littoral macrophytes as well as littoral sediment (benthic) habitats (thus 10 samples per waterbody). In 1995 and 1996, benthic and macrophyte samples were composited before sample processing while for 2006 and 2011, samples were separated and processed separately. Samples from littoral macrophytes were collected using standardised (10 sweeps per replicate sample) sweep-net sampling while samples from littoral sediments (<50 cm depth) were collected by sweeping the same pond nets over fixed areas of disturbed sediment (1995, 1996, 2006) or evacuating disturbed and suspended sediments from sampling cylinders placed over fixed, and enclosed and isolated, areas of sediment (2011). Water and sediment quality samples or data were gathered at the same time as the biological samples were taken.

Results for the four annual sampling occasions have been summarised and analysed using community summaries and multivariate statistical techniques. Community summaries reported here are based on taxa (usually family) number and total abundance. For this paper, the statistically-summarised data are compared amongst sites and years graphically. Multivariate analyses examine the relationships, including similarities, of communities (ie the relative abundances of constituent taxa) amongst sites and years. The multivariate analyses reported in this paper are multi-dimensional scaling ordination as well as tests of the separation of different community groups (eg from different mine-exposure categories) in multivariate space (ANOSIM), using routines from the PRIMER software package (Clarke & Gorley 2006).

Results

Water quality

Mine solutes input to GTB include magnesium sulfate and uranium. The concentrations of U in GTB have been generally low and importantly, in the antecedent periods (wet and dry seasons) leading up to sampling in 1995, 1996, 2006 and 2011, were at least an order of magnitude below the relevant trigger value for protection of local aquatic organisms (Hogan et al. 2010). Plots of (generally) weekly median EC values (a surrogate for $MgSO_4$) measured in GTB since 1991 are shown in Figure 1 in relation to wet season (January to May inclusive) and dry season (June to December) time periods.

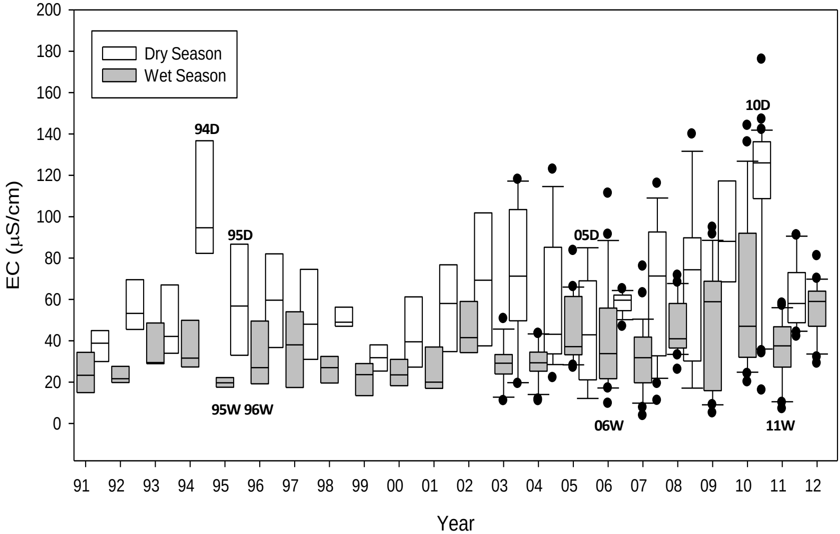


Figure 1 *Summary box plots of weekly electrical conductivity (EC) values measured in GTB between 1991 and 2012. Box plots show median, 25th and 75th percentiles, and points outside ±1.5 interquartile range. Periods relevant to macroinvertebrate sampling are indicated by year and season (wet (W) or dry (D)). Data from Energy Resources of Australia. Wet season – January to May. Dry season – June to December.*

The antecedent (ie the months prior to sampling) wet season median EC was low in 1995, 1996 and 2006 compared with the local EC trigger value (TV) of 42 µS/cm (based on an equivalent Mg TV of ~3 mg/L) for the protection of aquatic ecosystems (van Dam et al. 2010)). However, in 2011 and in the several wet season months prior to sampling, median EC values in GTB (Figure 1) approached the EC trigger value (weekly median of 2.5 mg/L Mg).

Antecedent dry season EC is typically much higher than wet season values in GTB due to evapo-concentration of solutes that occurs prior to significant runoff from the Ranger minesite and the rest of the GTB catchment. Prior to 1982, EC naturally reached median dry season values of 43 µS/cm but Mg was not a significant contributor to solute concentration (median value only 0.6 mg/L Mg) (ERA data). In relation to the four years in which macroinvertebrate sampling has occurred in GTB, dry season EC values in 1994 and 2010 (prior to 1995 and 2011 sampling, respectively) were unusually high (median dry season values: 1994, EC of 95 µS/cm and Mg 2.9 mg/L; 2010, EC of 128 µS/cm and Mg 8.8 mg/L). Jones et al. (2008) argued that antecedent wet season water quality, rather than antecedent dry season quality, would likely be of much greater significance to resident biota given (i) the recency of exposure, (ii) the much higher biological diversity present in waterbodies in the wet season months compared with the dry season, and (iii) that the local fauna would likely be more sensitive to inputs of solutes during the wet season given that the natural exposure condition for this period is water quality similar to background receiving waters which are characterised by extremely low solute concentrations.

Macroinvertebrate communities

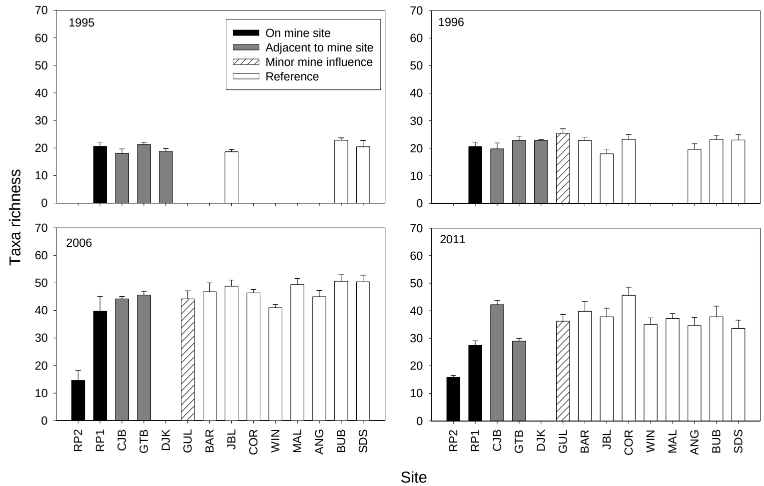
Apart from Ranger RP2, the mean taxa number and mean total abundance for macroinvertebrates sampled from macrophyte habitat type did not vary markedly amongst the waterbodies in 1995, 1996 and 2006 (Figure 2). For these years, in particular, the macroinvertebrate communities from GTB were very similar to those sampled from the same habitat type in reference waterbodies. In 2011, however, both mean taxa number and particularly mean total abundance in GTB were lower than values measured for reference waterbodies (Figure 2).

Multi-dimensional scaling ordination plots, used to depict the relationship of macroinvertebrate samples to one another, showed interspersion of replicate samples from GTB amongst reference waterbody samples in 1995, 1996 and 2006 but separation of GTB samples from reference waterbody samples in 2011. While these ordination plots are not shown here, the degree of separation of samples in multivariate space can be quantified using Analysis of Similarity (ANOSIM) – effectively an analogue of the univariate ANOVA (Table 1).

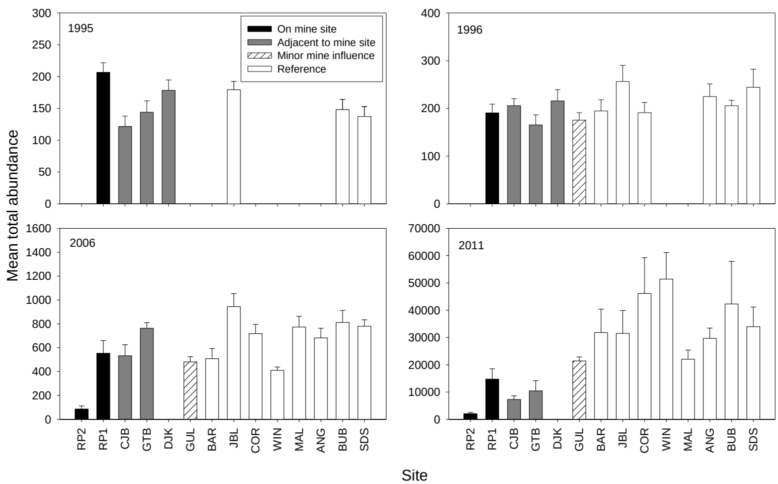
The ANOSIM test statistic compares the observed differences *between* groups (in this case exposure type, GTB versus reference waterbodies) with the differences amongst replicates *within* the groups. The test is based upon rank similarities between samples in the underlying Bray-Curtis similarity matrix. The degree of separation between groups is denoted by the R-statistic, where R-statistic > 0.75 = groups well separated, R-statistic > 0.5 = groups overlapping but clearly different, and R-statistic < 0.25 = groups barely separable. A significance level < 5% = significant effect/difference.

While GTB samples in 1995 were significantly different from reference waterbody samples, the ANOSIM R statistic is low and within the criterion from the technique's authors (Clarke & Gorley 2006) of 'barely separable'. On this basis, the macroinvertebrate communities of GTB in 1995 are not regarded as different from those in adjacent reference waterbodies in the region. For 2011, however, ANOSIM results indicate that GTB samples are clearly and significantly separated from reference waterbody samples. This result is consistent with community summary results reported above of reduced diversity of GTB macroinvertebrate communities compared to reference waterbody communities.

In summary, results from 2011 macroinvertebrate sampling indicate that the biological condition of GTB was no longer consistent with conditions occurring in reference waterbodies. This is coincident with an increase in solute concentrations observed during the antecedent dry and wet seasons. The significant, but not marked, separation of GTB samples from reference waterbody samples in 1995 coincides with poor antecedent dry season water quality in GTB (elevated EC and Mg).



a)



b)

Figure 2 *Histograms of mean (±SE) (a) taxa number and (b) macroinvertebrate abundance among waterbodies on or near the Ranger uranium mine site. Histogram shading depicts the gradient of exposure to mine waters, from most contaminated ('on mine site') to no exposure ('reference'). Site codes are Ranger Retention Pond 2 (RP2) and Retention Pond 1 (RP1), Coonjimba (CJB), Georgetown (GTB), Djalkmara (DJK), Gulungul (GUL), Baralil (BAR), Jabiru Lake (JBL), Corndorl (COR), Wirnmuyurr (WIN), Malabanjbanjdju (MAL), Anbangbang (ANG), Buba (BUB) and Sandy (SDS) Billabongs.*

Table 1 *ANOSIM summary statistic results for each year comparing GTB with reference waterbodies (no mine influence).*

Year	R Statistic	Significance Level %
1995	0.296	1.7
1996	-0.064	65.8
2006	0.07	30.8
2011	0.594	0.1

Conclusions

Macroinvertebrate sampling of waterbodies in 2011, in and around the Ranger mine lease, revealed for the first time, evidence of water quality impacts upon the fauna of GTB, coincident with poorer water quality observed in the immediately-prior wet and dry seasons. The EC and Mg concentrations observed in these antecedent periods were consistent with those for which biological effects may occur based upon trigger values produced by laboratory ecotoxicity studies. The ("no effects") data sets from 1995, 1996 and 2006 will therefore continue to underpin the setting of water quality closure criteria for U, Mg and EC.

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