Assessing the ecological impact of water abstraction on macroinvertebrates in unregulated rivers – reference site selection

Leading policy and reform in sustainable water management

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Summary

Nineteen unregulated river water sharing plans prepared under the New South Wales (NSW) *Water Management Act 2000* came into effect on 1 July 2004. The purpose of these plans is to provide water to sustain aquatic environments while defining an allowable level of water extraction and procedures to share extracted water among users.

Water sharing plans have statutory status and include a generic ecological performance indicator, expressed as follows: 'change in ecological condition of this water source and dependent ecosystems as measured by periodic assessment of identified attributes of this water source and dependent ecosystems.' Accordingly, there is a legal requirement for ecological monitoring and reporting in each plan area.

An assessment of the biological impact of water abstractions in unregulated streams is required to distinguish between the effects of hydrological alteration and confounding factors. This will assist the understanding of the extent and nature of the impacts of current water abstraction, and will enable the assessment of any changes to the biota under different water management rules. Fundamental to any assessment is the selection of reference condition information to define water management goals, determine the restoration potential of sites and evaluate the success of water management actions.

This report describes a method for selecting reference sites for biological assessment using aquatic macroinvertebrates. The assessment method takes a set of reference sites and selects those that are most appropriate for comparison with a particular exposure site by considering multiple environmental factors that may limit the maximum similarity of biological assemblages among sites.

This method was used to select reference sites for the assessment of the impact of water abstraction on macroinvertebrate assemblages at five priority unregulated water sharing plan areas. The high priority plan areas were Kangaroo River Water Source, Dorrigo Plateau Surface Water Source, Karuah River Water Source (Zone 1), Upper Coopers Creek Management Zone and Tenterfield Creek Water Source (Zone 5). These plans were selected on the basis of high environmental risk and high to medium community dependence as outlined in Raine *et al.* (2009).

Between four and 150 reference sites were chosen for each plan site. It is not feasible to sample up to 150 sites in an assessment program therefore the total number of reference sites was reduced to 11 by selecting only sites that could be used as reference conditions for multiple plan areas. The reference sites selected will form the basis of the first stage of assessing the ecological outcomes of the high priority water sharing plans for unregulated water sources. All five plan sites and reference sites will be sampled for macroinvertebrates twice per year (autumn and spring) beginning in spring 2009. Macroinvertebrates will be sampled at each site from riffle habitats using both quantitative and qualitative methods.

Background

Thirty-one water sharing plans prepared under the New South Wales *Water Management Act 2000* came into effect on 1 July 2004. These comprise 19 unregulated river plans, 1 combined unregulated river and groundwater plans, 7 regulated river plans and 4 groundwater plans. The purpose of these plans is to provide water to sustain aquatic environments while defining an allowable level of water extraction and procedures to share extracted water among users. As required by section 35 (1) of the Act, these plans include a vision statement, objectives consistent with the vision statement, strategies for reaching those objectives and performance indicators to measure the success of those strategies. Additional 'macro' water sharing plans, covering whole river basins, are now in preparation and are expected to include most of the remaining unregulated rivers in the State. More detailed plans may also be prepared for particular water bodies where a higher level of management is required.

The plans have statutory status, include performance indicators to determine the performance of each plan against its objectives, and require that monitoring and reporting (M & R) of these performance indicators be undertaken. All plans include a generic ecological performance indicator, expressed as follows: 'change in ecological condition of this water source and dependent ecosystems as measured by periodic assessment of identified attributes of this water source and dependent ecosystems.' Accordingly, there is a legal requirement for ecological M & R in each plan area (although not necessarily in every water source).

In accordance with section 35 (1) of the *Water Management Act 2000*, the purpose of performance indicators developed for plans is to measure the success of plan strategies in achieving plan objectives. It follows that ecological M & R must be able to answer the following questions:

1. Were the ecological objectives of the plan achieved?

2. How was achievement (or non-achievement) of these objectives related to plan strategies?

3. Are the ecological objectives of the plan appropriate?

The second question is required because it is possible that plan objectives could be achieved, or could fail to be achieved, because of factors outside the ambit of plans. Such factors might include climate change or non-hydrological influences on aquatic ecosystems such as wildfires and changes in land use.

Introduction

The hydrological and ecological impacts of flow modification on riverine ecosystems have been widely studied (Petts 1984; Pringle *et al.* 2000; Magilligan and Nislow 2005). Altered flow regimes affect macroinvertebrates and other aquatic fauna by altering physical habitat, interrupting their life history, limiting or increasing longitudinal and lateral connectivity depending on the nature of the alteration, and facilitating the invasion and success of introduced species (Bunn and Arthington 2002). Macroinvertebrate assemblages may also be affected by changes to water chemistry caused by flow alterations such as reduced dissolved oxygen levels (McKay and King 2006) and increased water temperatures (Rader and Belish 1999). A number of studies have demonstrated changes to macroinvertebrate assemblages due to alterations of flow regimes caused by dams (Boon 1988; Armitage and Pardo 1995; Englund and Malvquist 1996; Pringle *et al.* 2000; Miller *et al.* 2007) but there are few that have attempted to assess the impact of water abstraction from unregulated streams.

In Australia, water is pumped from streams with unregulated flow for town water supplies, irrigation, farmstead use, livestock watering and other purposes (Reinfelds *et al.* 2006) and agriculture accounts for approximately 65% of all water use in Australia (ABS 2006). In south eastern Australia, the greatest level of water diversions coincides with natural periods of low flow, generally during summer (November to March). Reinfelds *et al.* (2006) has shown that in the Bega River up to 91% of daily flows are diverted for irrigation during low flows, however total annual diversions only comprises 7% of mean annual natural flow, highlighting the temporal nature of this impact. Moreover, the amount of irrigation during low flows also varies from year to year. These water abstractions cause an increase in the frequency and duration of low flows, and accentuate the effects of drought. In addition, the effects of irrigation diversions on daily flows are not uniform from year to year and are most pronounced during drought periods. The hydraulic response to water abstraction is most pronounced in running water mesohabitats such as riffles, where wetted area, average depth and velocity are greatly reduced under low flow conditions (Reinfelds *et al.* 2004; Dewson *et al.* 2007a). The hydraulic changes can alter macroinvertebrate assemblages directly through the loss of an organism's preferred habitat and a reduction in overall habitat diversity (Stanley *et al.* 1997). Macroinvertebrates can also be indirectly affected through changes to water chemistry associated with extended periods of low flows such as increases in water temperature and conductivity and decreases in dissolved oxygen (Boulton 2003; Miller *et al*. 2007). Changes to macroinvertebrate assemblages have also been found to be related to increased periphyton biomass during a period of summer low flows (Suren *et al.* 2003).

There have been few studies that have attempted to quantify the ecological impact of water abstraction on macroinvertebrates in unregulated streams. Both observational (Rader and Belish 1999; McIntosh *et al.* 2002; Suren *et al*. 2003) and experimental studies (McKay and King 2006; Wills *et al*. 2006; Dewson *et al.* 2007a) have produced inconsistent results. The inconsistencies probably result from differences in the duration and degree of hydrologic alteration between studies and the type of study itself (observational or experimental). Experimental or manipulative studies are an important component in understanding flow-ecological relationships (Bunn and Arthington 2002). However, they may be limited in their ability to mimic impacts of water abstraction because they are generally undertaken over short time periods (weeks to months) and do not allow for potential lag effects in the biological responses. The limited spatial extent of experiments may not truly reflect the possible impact of water diversions on recolonisation processes. Observational studies generally suffer from a lack of pre-water abstraction information and have difficulty in separating the direct effects of modified flow regime from confounding land-use impacts (Bunn and Arthington 2002; Dewson *et al.* 2007b).

An assessment of the biological impact of water abstractions in unregulated streams is required to be able to distinguish between the effects of hydrological alteration and confounding factors. This will assist the understanding of the extent and nature of the impacts of current water abstraction, and will enable the assessment of any changes to the biota under different water management rules. Fundamental to any assessment is the selection of reference information to define water management goals, determine the restoration potential of sites and evaluate the success of water management actions (White and Walker 1997). Chessman, Muschal and Royal (2008a, 2008b) have recently described a method to compare biological assemblages between a stream reach exposed to water abstraction (an exposure reach) and reference reaches with little, or preferably no, water diversions, while taking account of the role of other environmental factors that are associated with spatial variation in the assemblage. The method, named the limiting environmental difference (LED) approach, takes a set of reference sites and selects those that are most appropriate for comparison with a particular exposure site. This is done by considering multiple environmental factors that may limit the maximum similarity of biological assemblages among sites (Thomson *et al.* 1996; Lancaster and Belyea 2006).

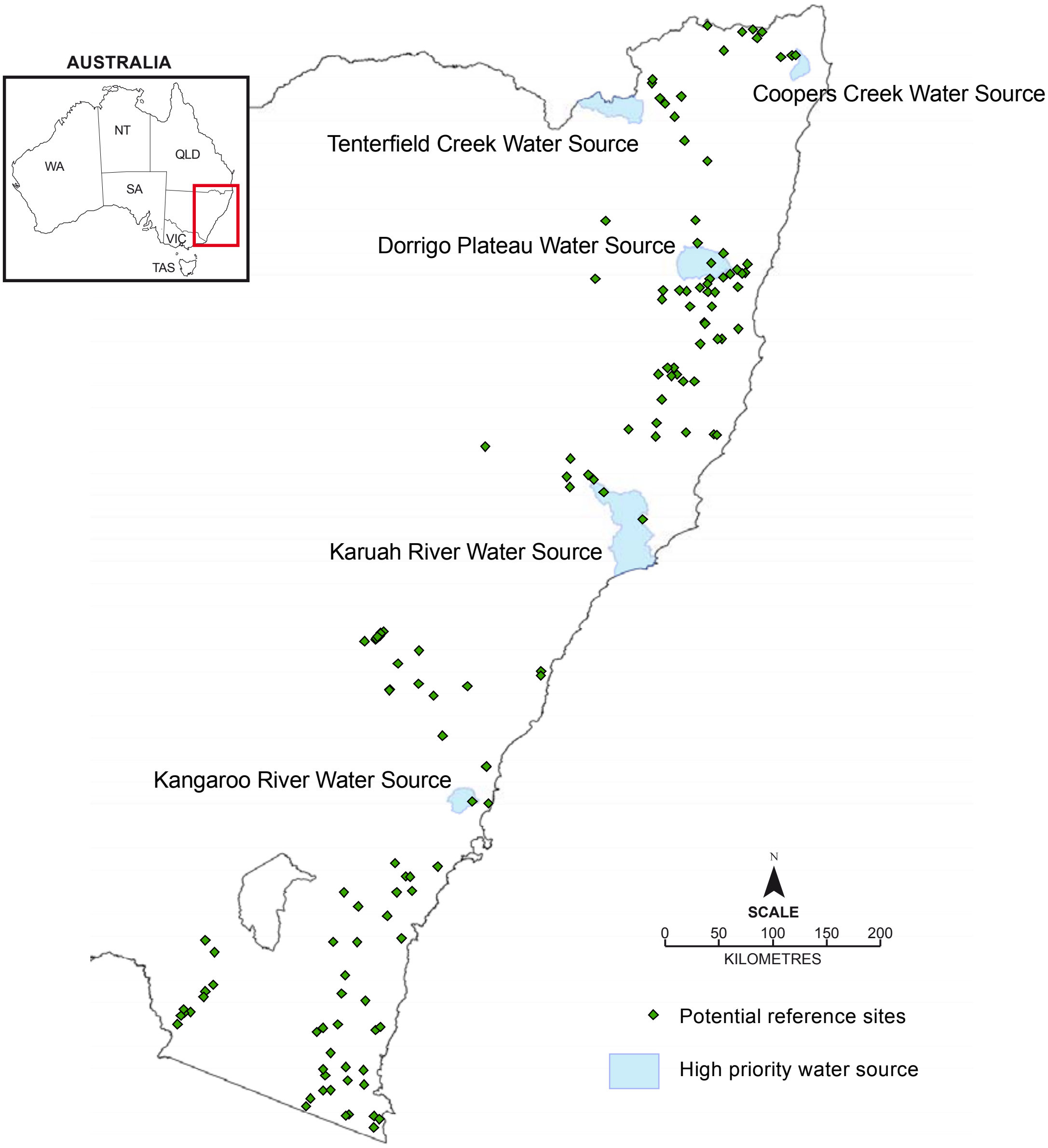
In this study we have used the LED method with some modifications to select reference sites for the assessment of the impact of water abstraction on macroinvertebrate assemblages at five unregulated plan areas. The plan areas that we selected reference sites for were Kangaroo River Water Source, Dorrigo Plateau Surface Water Source, Karuah River Water Source (Zone 1), Upper Coopers Creek Management Zone and Tenterfield Creek Water Source (Zone 5). These plans were selected on the basis of high environmental risk and high to medium community dependence as outlined in Raine *et al.* (2009).

Methods

Study area

The study was conducted in New South Wales, across a mix of land uses ranging from national parks to sheep and cattle grazing, irrigated dairy farms and urban settlements. The study area is considered to be temperate, and the Great Dividing Range, running approximately north to south in the east of New South Wales, has a large impact on the climate, creating four distinct climate zones; the coastal strip, the highlands, the Western Slopes and the flatter country to the west (Bureau of Meteorology 2009). Sites were sampled from streams draining both east of the Great Dividing Range to the Pacific Ocean and west of the range to the Murray-Darling Basin (Figure 1).

Figure 1. Location of high priority water sharing plan areas and potential reference sites.



Macroinvertebrate sampling

In this study we used pre-existing data that was collected by others as part of the National River Health Program. Macroinvertebrates were sampled from 1995-2000 at 305 sites from riffle habitats in unregulated streams throughout New South Wales (Figure 1), as part of the National River Health

Program (see Turak *et al.* 1999 and Turak *et al.* 2000). Macroinvertebrates were collected using the rapid assessment methods described by Turak and Waddell (2004). Not all sites were sampled every year and only macroinvertebrate samples collected from riffle habitats in autumn were used in the analyses. This habitat and time of year was chosen as it is the most likely to exhibit evidence of biological stress due to water demands over summer. No sites below major water storages were selected as the focus of the study was on unregulated streams.

Classification of reference sites

Reference sites were considered to be streams with values of hydrological stress = 0. Hydrological stress was computed by expressing the upstream water allocation as a percentage of the modelled natural flow exceeded 90% of the time (Stein, Hutchison and Stein 2007). Reference sites were not necessarily pristine and could be affected by other human influences, the only criterion for this classification were that there was an absence of hydrological stress.

Environmental data

A range of environmental data was used as possible predictors of macroinvertebrate assemblage composition (Table 1). Eleven major landuse categories were extracted from the NSW GIS landuse layer (Table 1). The percentage of each of the 11 major landuse categories for each catchment draining to the sampling point was included as a possible predictor variable. At the terminus of each plan area's major water source, a site was chosen for which all environmental variables were calculated.

Derivation of Maximum Allowable Environmental Difference (MAED) values and predictive models

All analyses using macroinvertebrate data were carried out using family level binary (presence/absence) data.

Upper limiting relationships between the similarity of macroinvertebrate assemblages from reference sites and differences in individual environmental variables were determined. This was carried out to define criteria with which to select a subset of reference sites for comparison with each plan's site. Similarities of macroinvertebrate assemblages between the two members of each possible pair of reference sites were calculated using the Bray-Curtis (BC) similarity measure. Environmental differences were calculated as the absolute value of the difference in each environmental variable between the two members of a pair.

Upper limiting relationships between the similarity of macroinvertebrate assemblages and environmental differences were investigated for all reference-site pairs and each environmental variable using quantile regression. Quantile regression is a statistical technique that is well suited to estimating the effects of limiting factors (Cade, Terrell and Schroeder 1999). The quantile regression algorithm finds a function where the proportion of observations equal to the specified quantile τ (ranging between 0 and 1) will fall at or below the regression line (Cade and Noon 2003; Lancaster and Belyea 2006). For example, for $\tau = 0.90$, 90% of the values of the response variable will be less than or equal to the quantile regression function. We calculated quantile regression functions to describe the upper limiting ($\tau = 0.99$) relationships between the differences in environmental variables and BC similarity of macroinvertebrate assemblages in reference sites. Two models of ecological response (log-linear and power models) were fitted to the data for each variable and the best model was used in the LED method.

Quantile regression was carried out using the quantreg package in R Project software (R Development Core Team 2009). The models were fitted using the modified version of the Barrodale and Roberts algorithm for l_1-regression, and t- and p-values for the estimated coefficients were calculated using a Huber sandwich formula and the Hall-Sheather bandwidth rule (Koenker and d'Orey 1987; Koenker and d'Orey 1994). To select the best model for each environmental variable for use in the derivation of MAEDs we chose statistically significant models with R^1 (coefficient of determination) values greater than 0.01 (Appendix C in Cade, Noon and Flather 2005).

For each significant upper limiting environment-similarity relationship, an environmental difference value was calculated for each BC similarity threshold and was used to define the maximum allowable environmental difference (MAED) between a WSP sites and potential reference sites. MAEDs were calculated for a range of BC similarity thresholds (0.70 to 0.75) and a “null” model where all reference sites were included (BC threshold = 0). This was done to test the effect of varying the similarity threshold on the accuracy of the prediction.

Selection of reference sites for assessment of the impact of water abstraction on macroinvertebrate assemblages in plan areas using MAED predictive models

The MAEDs were used to select a subset of reference sites to be compared with each plan’s site in a future assessment program. For each environmental variable, the difference between a potential reference site and each plan site was calculated. Only reference sites for which every environmental difference was less than the MAED were selected for comparison with that WSP site. The sub-set of reference sites was generally unique for each plan site due to differences in environmental variables amongst sites. Each reference site was compared with a sub-set of other reference sites using the Bray-Curtis similarity measure to assess the predictive accuracy and precision. No reference site was allowed to be included in the sub-set of references to avoid comparison with itself.

Accuracy of the predictive model for each BC threshold was assessed as the proximity of the median BC similarity between predicted reference site assemblages and observed reference site assemblages to a value of 1. A median BC value of 1 indicates that the MAED method predicts exactly the macroinvertebrate assemblage at a given reference site. Ideally, reference site predictions would match the observed reference site assemblages perfectly (i.e. a median BC value of 1), but in reality this is not likely to occur because of sampling variation and the MAED model with the highest median BC similarity value was considered to be the most accurate.

All predictive models (BC cutoff 0.70 to 0.75) were then used to obtain reference sites for the five plan areas. The most accurate model in which at least four reference sites qualified for comparison with each plan site was selected as the most appropriate model.

Results and discussion

Derivation of MAEDs and predictive models

One hundred and thirty-eight sites were classified as reference sites. Table 1 lists the median and ranges for the environmental variables used in the study. The median hydrological stress for the non-reference sites was 22.2%. One-hundred and sixty eight families of macroinvertebrates were recorded from the 305 sites.

Table 1. Median and ranges of environmental variables for reference sites and MAED values for selected 72.5 Bray-Curtis similarity threshold. Only MAEDs for significant upper limiting relationships are shown. MAED values were not calculated for hydrological stress.

Variable	**Reference** (n=217)	**MAED** BC cutoff=72.5
Hydrological stress (%)	0	
Conservation area (proportion of catchment)	0.99 (0 – 1.00)	063.*
Cropping (proportion of catchment)	0 (0 – 0.07)	
Grazing (proportion of catchment)	0 (0 – 0.98)	0.26*
Horticulture (proportion of catchment)	0 (0 – 0.02)	
Mining & quarrying (proportion of catchment)	0 (0 – 0.15)	0.007*
Power generation (proportion of catchment)	0 (0 – 0.10)	
River & drainage (proportion of catchment)	0 (0 – 0.04)	
Transport (proportion of catchment)	0.006 (0 – 0.980)	0.67*
Tree & shrub cover (proportion of catchment)	0 (0 – 1.0)	
Urban (proportion of catchment)	0 (0 – 0.45)	
Wetland (proportion of catchment)	0 (0 – 0.007)	
Lattitude (decimal degrees)	-33.3824 (-37.3752 – -28.3270)	7.18147*
Longitude (decimal degrees)	150.4023 (148.2644 – 153.3425)	3.34290*
Catchment area (km^2)	20.2 (1.5 – 357.5)	450.8
Average 6 month (October-March) rainfall (mm)	653.3 (396.2 – 1339.2)	1826.0
Altitude (m)	310 (0 – 1920)	806.7

* based on log-linear model

Significant upper limiting environment-BC similarity relationships for pairs of reference sites were found for seven environmental variables (Table 1, Figure 2).

As the MAED selection criteria increased (lower BC thresholds), the number of reference sites qualifying for comparison with each observed reference site increased, and the models became increasingly inaccurate (Figure 3). The null model, which included all reference sites, was the least accurate (Figure 3).

Figure 2. Examples of relationships between BC similarity of reference macroinvertebrate assemblages and differences in values of environmental variables for all pairs of reference sites. The regression line is the upper limiting quantile regression function (τ=0.99) and used in the calculation of MAEDs.

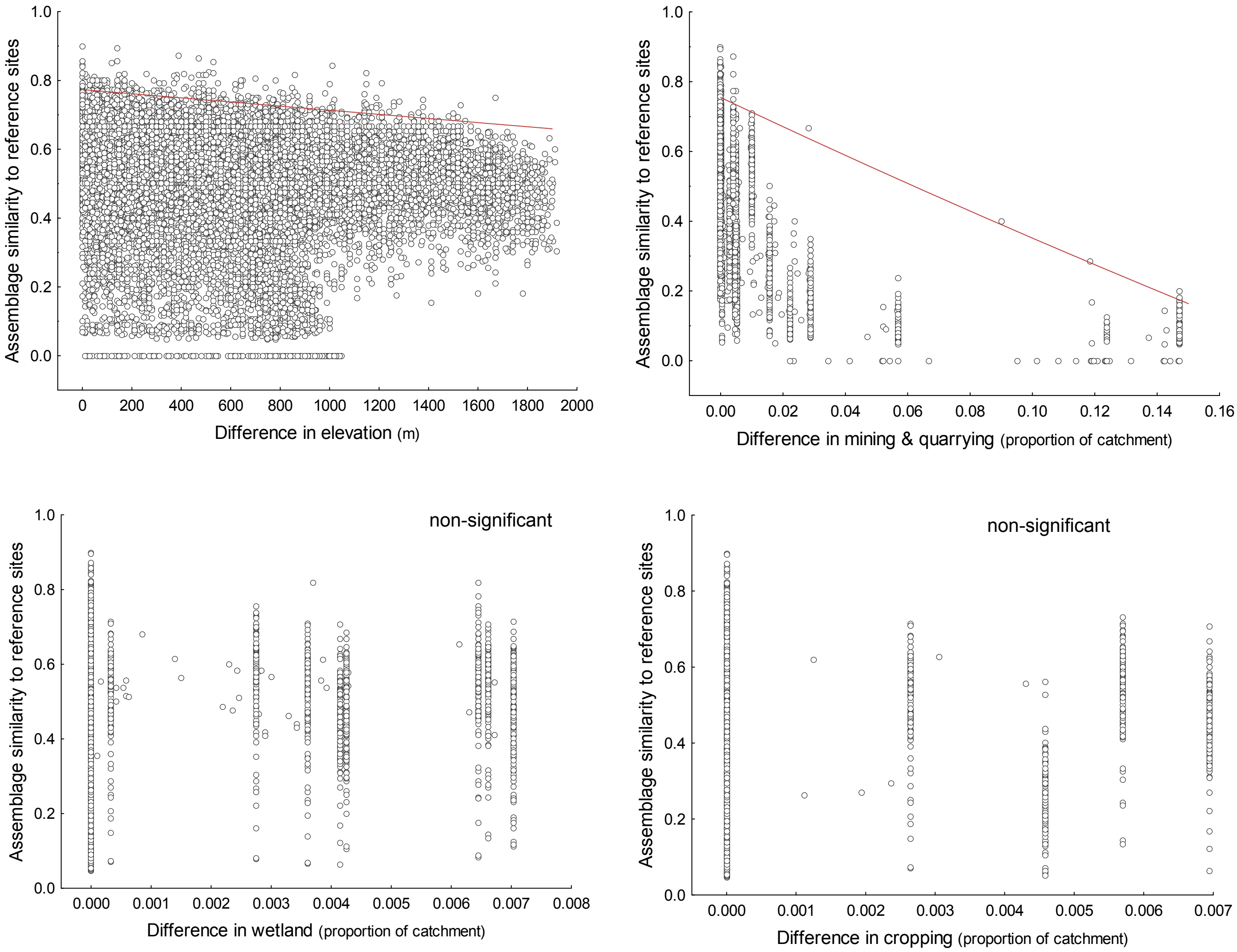
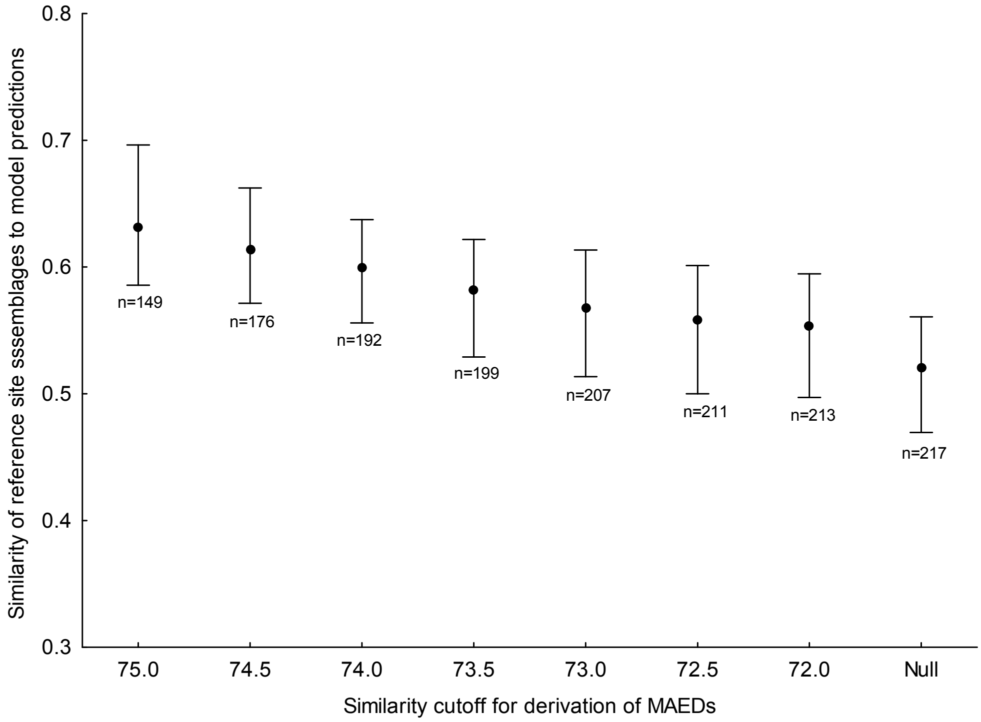


Figure 3. Comparison of accuracy of macroinvertebrate assemblage predictions for different similarity thresholds. The median and inter-quartile range of BC similarities was calculated between reference site assemblages and assemblages predicted by the LED model.



Reference site selection

The most accurate model in which most plan areas had four or more qualifying reference sites was the 0.725 BC threshold (Figure 3). Both the Tenterfield Creek Water Source (Zone 5) and Dorrigo Plateau Surface Water Source required adjustment of the catchment area MAED (increased to 805 km^2 and 950 km^2 respectively) to allow sufficient number of reference sites to be selected. Using the LED model and adjusted model, between 4 and 150 reference sites were chosen for each plan site. It is not feasible to sample up to 150 sites in an assessment program therefore the total number of reference sites was reduced to 11 by selecting only sites that could be used as reference conditions for multiple plan areas. Table 2 summarises the results of the reference site selection process. Appendix 1 shows the geographical position of reference sites for each plan area.

Table 2. Reference sites selected for comparison with five water sharing plan areas.

Dorrigo Plateau Surface Water Source	Kangaroo River Water Source	Karuah River Water Source - Zone 1	Upper Coopers Creek Management Zone	Tenterfield Creek Water Source - Zone 5
BEGA11	BELL584	BELL584	BELL584	BELL702
BELL702	CLAR04	CLAR04	CLAR04	CLAR05
CLAR05	CLAR104	CLAR104	CLAR104	RICH540
RICH540	CLAR18	CLAR18	CLAR18	SHOA572
SHOA572	MACL576	MACL576	MACL576	
	MANN577	MANN577	MANN577	
	RICH540		RICH540	
	SHOA572		SHOA572	

Proposed program for ecological assessment of water sharing plans

The reference sites selected above will form the basis of the first stage of assessing the ecological outcomes of the high priority unregulated water sharing plans. All five plan sites and 11 reference sites will be sampled for macroinvertebrates twice per year (autumn and spring) beginning in spring 2009. Macroinvertebrates will be sampled at each site from riffle habitats using both quantitative and qualitative methods. For the quantitative sampling, macroinvertebrates will be sampled from ten random points in a single riffle at each site. A suction sampler described by Brooks (1994) will be placed over the substrate and operated for one minute at each sampling location. The sample is to be washed thoroughly through a 250µm mesh sieve and matter retained on the sieve will be preserved in a jar of 70% ethanol. Qualitative sampling will be undertaken in riffles using the methods outlined in Turak and Waddell (2004). All invertebrates will be identified to Genus level.

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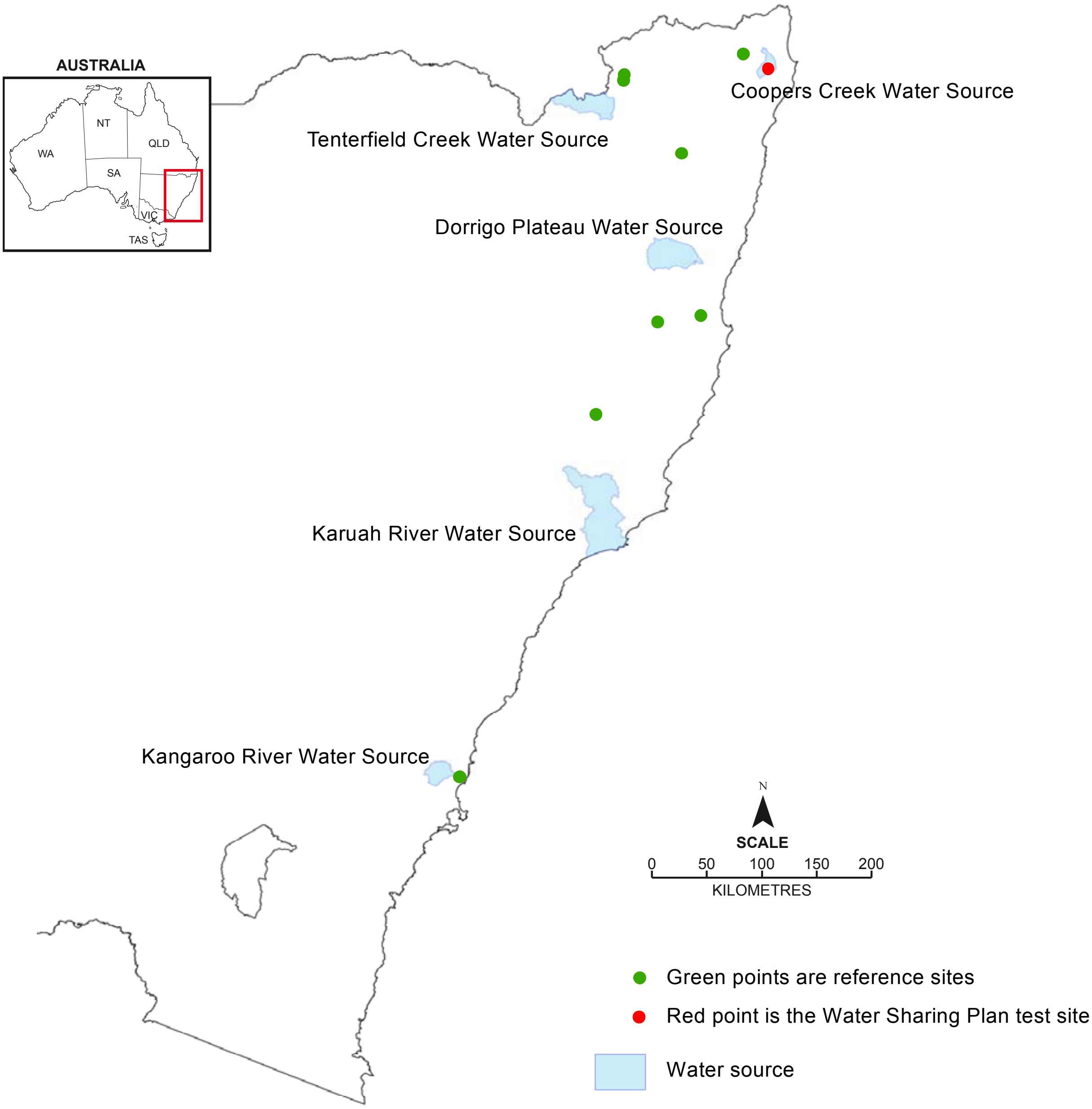
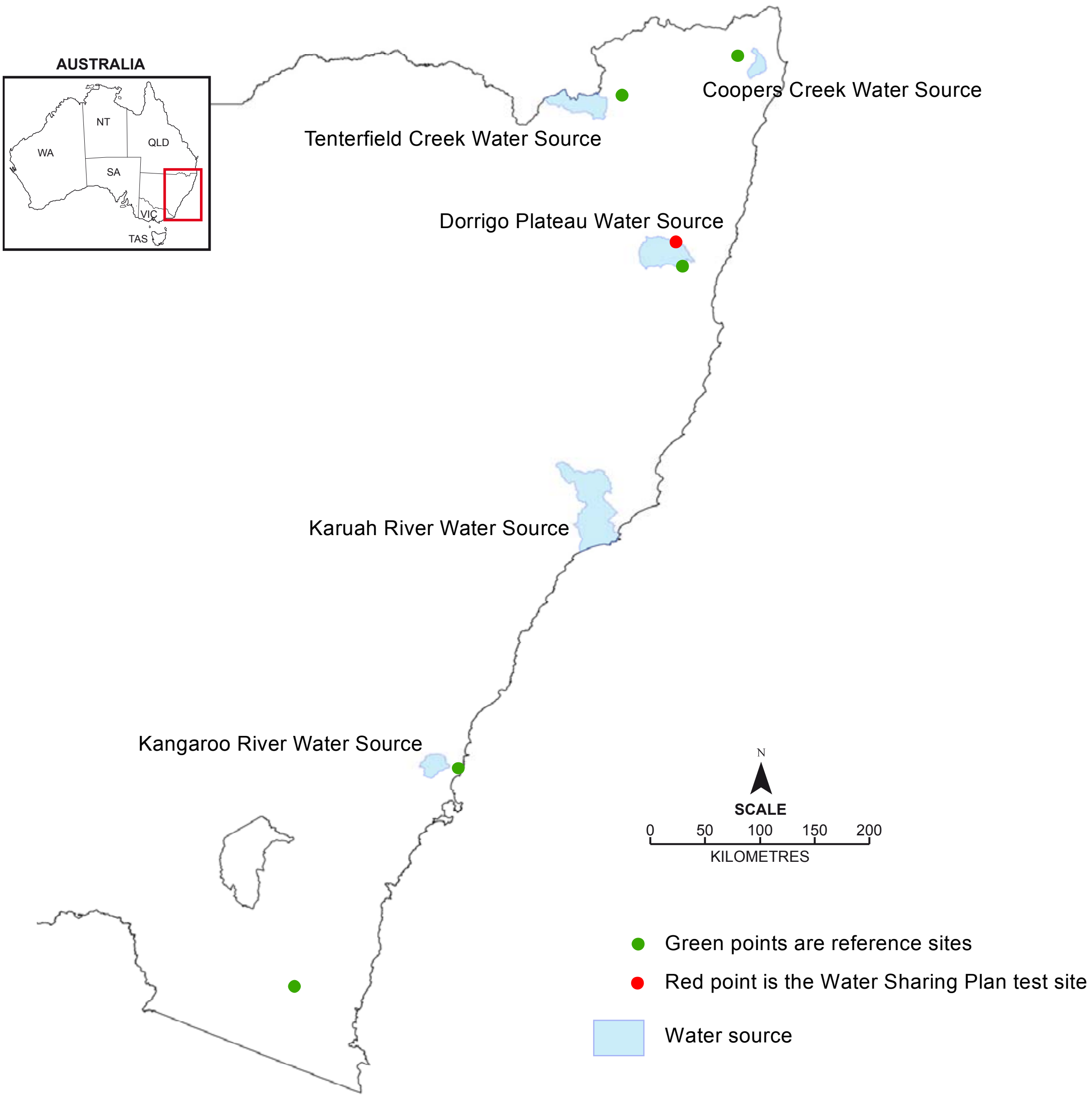
Appendix 1

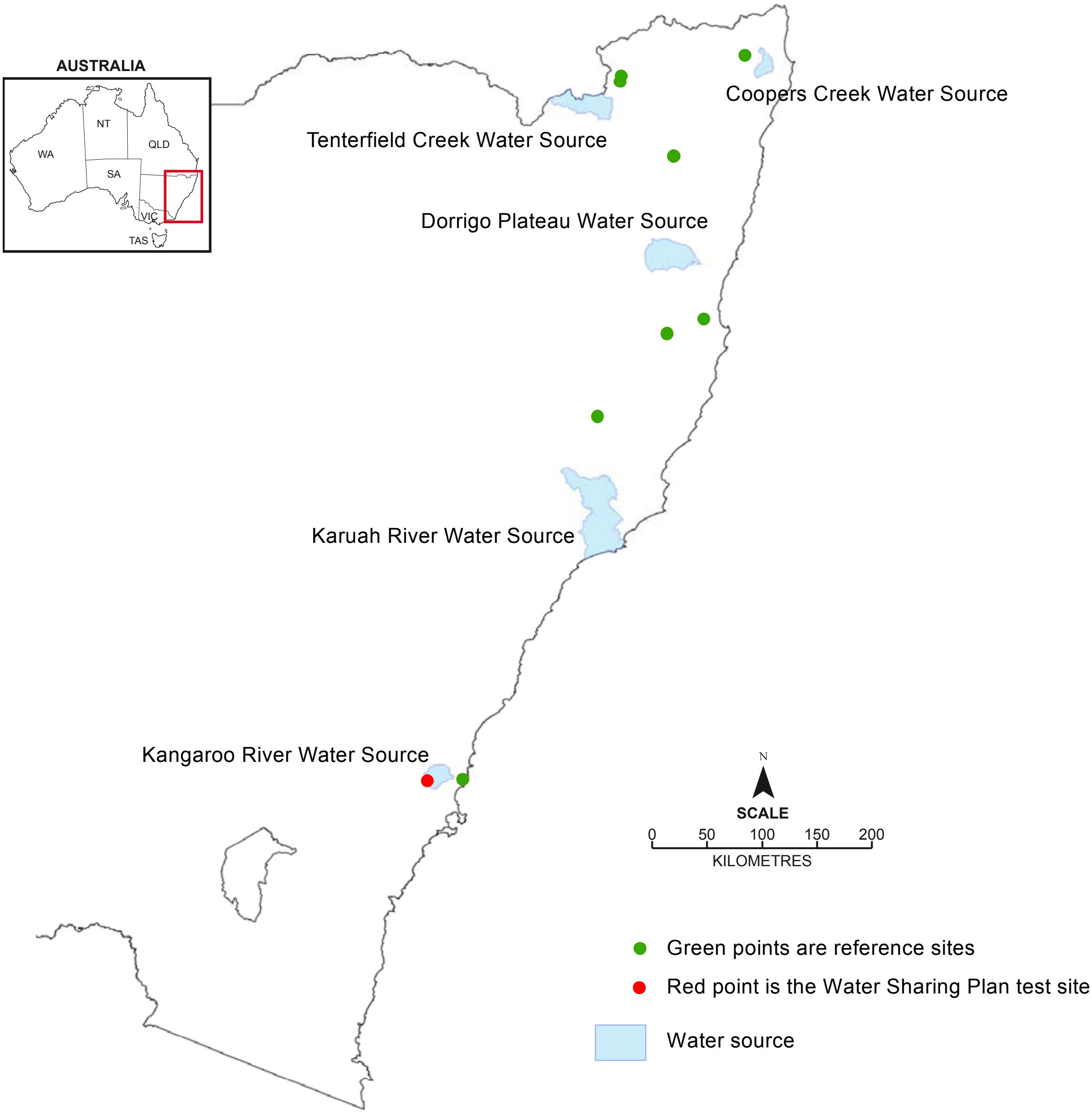
Macroinvertebrate sampling locations for Coopers Creek Water Source.



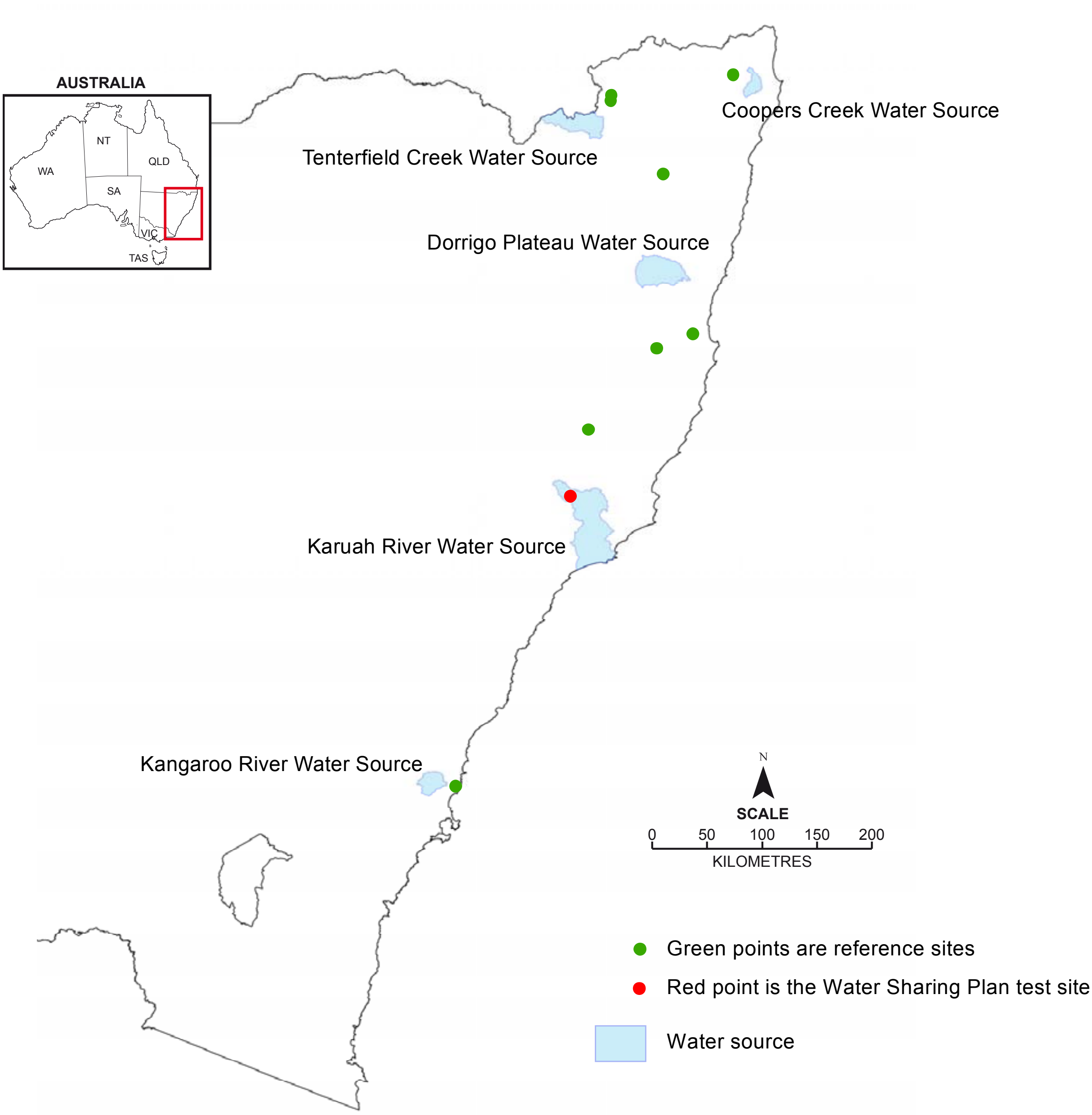
Macroinvertebrate sampling locations for Dorrigo Plateau Water Source.



Macroinvertebrate sampling locations for Kangaroo River Water Source.



Macroinvertebrate sampling locations for Karuah River Water Source.



Macroinvertebrate sampling locations for Tenterfield Creek Water Source.