A Compendium of Ecological Information on Australia's Northern Tropical Rivers

# REPORT 5

# Water Quality


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## Disclaimer

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# 1. INTRODUCTION

The terms of reference for this, the water quality component of the Tropical Rivers Inventory and Assessment Project (TRIAP), called for investigators to determine the feasibility of developing a water quality-based geospatial classification scheme capable of differentiating riverine habitats that perform inherently distinctive ecological functions. There are no real precedents for a typology of this kind, so it was appreciated from the outset that this was an ambitious undertaking and that there was little prospect of developing a working classification scheme within the lifetime of this project. The main aim of this study was to lay the groundwork for the future by accomplishing the following objectives:

- Identify the key biophysical processes that govern relationships between water quality and the ecological functioning of northern river systems;
- Determine if there are indicator parameters and/or monitoring methods that could potentially be used to assess the variability of these natural processes, over local, regional and/or continental scales, and;
- Explain how the data generated by such assessments provide an ecologically meaningful basis for reach-scale classification of river systems.

During the early stages of the study it was concluded that available data were far too limited to provide an adequate basis for the development of an empirical classification scheme, especially one that is truly ecologically meaningful. It is currently possible to identify statistically distinctive water quality types within available datasets (see for example Horrigan *et al*, 2005); however, due to existing deficiencies in spatio- temporal coverages (detailed in the appendix), and the limited range of indicator parameters that have been monitored, water quality types identified through this means are highly unlikely to be strongly or directly correlated to ecological function, and are of no value to the current project. This finding must be taken in context – the available data were collected by numerous unrelated projects, none of which were necessarily designed to answer any of the questions being posed in this study. The numerous data deficiencies discussed throughout this report relate only to the capacity of the data to serve this unintended purpose and should not be taken as an indictment of the investigations that gathered the data.

The data available to this study represent only a small proportion of the relevant information that has been collected in the study area over the years. Many detailed studies of the water quality and/or ecology of northern rivers have been carried out in connection with a variety of research and consultancy projects but, due to issues such as client confidentiality and intellectual property rights, a lot of these data may never be publicly accessible. Moreover, owing to the lack of any coordinated or standarised system of storing the information produced by studies of this kind, it would be an enormous undertaking to encode the data in a form suitable for analysis.

However, a large proportion of the studies in question have been conducted either by members of the TRIAP study team or their colleagues, and the knowledge gained in the process has provided some unique perspectives on how local river systems work. This study sought to exploit this extensive pool of experience by convening a series of in-house workshops to exchange professional opinions about the roles that water quality and related biophysical factors play in determining the fundamental ecology of northern rivers. The conceptual understanding gained through this process is summarised in Section 2, which details the ways in which the interactions between natural hydrological and limnological processes and water quality variables are thought to influence the spatio-temporal distribution of ecological functions within northern river systems. These concepts, which underpin subsequent discussions in this report, provide a basis for suggesting some heuristic classification methods that could be applied to northern river systems in the future. These are discussed in Sections 4 and 5.

## 1.1 Summary of Findings

The expert panel concluded that, although virtually all water quality parameters play some supporting role, the natural ecology of most freshwater habitats in northern Australia is fundamentally governed by two overarching water quality related variables, namely optical depth and dissolved oxygen status. Salinity, or more specifically salinity regime, is also a fundamentally important determinant of ecosystem type and function for all waters that are potentially subject to marine influences, but it is a much more secondary concern in freshwater systems.

Salinity, and related ionic constituents such as alkalinity, hardness and chloride, cannot be ignored in ecological investigations as they modify the composition of freshwater communities, and can also significantly alter the ecosystem's susceptibility to adverse effects from other water quality parameters. However, as Section 3 explains, in this part of Australia natural concentration ranges are too narrow for these parameters to be deterministic or indicative of functional diversity, especially at broad regional spatial scales.

For reasons that are explained in Section 2.2, the majority of freshwater habitats in northern Australia experience hypoxic (i.e. oxygen deficient) conditions at some stage of the year. Since most aquatic organisms can rapidly asphyxiate if dissolved oxygen (DO) concentrations get too low, annual ecological outcomes for many biological communities can hinge on whether or not they survive a single brief hypoxia event. Episodes of this kind are most likely to occur during the pre-wet season, and are often precipitated by pre-flush storm events – i.e. rain events that deliver runoff to streams without generating sufficient flow to flush them out (see Section 2.1). Even under normal dry season conditions, most waterbodies experience daily DO sags and many contain water layers and benthic habitats that are severely hypoxic most of the time (see Section 2.2). Consequently, habitat utilisation, productivity and survival can be so strongly governed by oxygen availability that it is rarely possible to fully understand the ecology or condition of a waterbody without taking its DO status into consideration.

The DO status of a lentic and/or slowly flowing water is largely dependent on its limnology and metabolism (i.e. the existing balance between photosynthesis and respiration processes). Hence DO is strongly influenced by any factors that alter the type and amount of productivity occurring within a body of water, and this makes it a sensitive indicator of the net ecological effects resulting from interactions between other key ecological water quality variables such as nutrients, turbidity, colour and organic carbon. Unfortunately the spatio-temporal variability that makes DO such an important determinant and indicator of ecological condition also makes it one of the most difficult parameters to monitor. Until very recently data interpretation capabilities have also been severely constrained by a lack knowledge of the DO requirements of local species, and so, DO has largely been neglected as an indicator for routine monitoring applications and ecological assessments. (Some databases contain significant numbers of spot measurements, but as can be seen in Section 2, these are not interpretable).

These problems have attracted considerable research attention in recent years and as a result DO guidelines are now available for the freshwater ecosystems of northern Australia (Butler and Burrows, in review). These provide detailed information about the relative hypoxia sensitivities of many northern freshwater species and propose some new methods for assessing hypoxia risks and monitoring DO. The new methods make it feasible to identify high risk sites during the course of routine monitoring programs, and thus provide a means of ensuring that more intensive monitoring efforts focus on high priority sites. However, the task of accurately determining the DO status of aquatic habitats remains a significant undertaking, and although the new methods considerably streamline the process and provide considerably enhanced interpretation capabilities, work of this kind will almost certainly only be carried out at high priority sites. Since DO status can vary enormously, even between sites located within the same river reach, it is highly unlikely that spatial coverages will ever be adequate for DO data to be an effective tool for broad-scale river classification.

However, the factors that influence DO status are well understood and predictive capabilities are rapidly improving to the point where it may soon be possible to develop a classification scheme based on the primary biophysical drivers of DO status rather than DO measurements *per se*. Notably, because they carry little or no flow, most rivers in this region are poorly aerated for most of the year so water oxygenation is heavily dependent on instream aquatic autotrophs photosynthetically producing sufficient oxygen to compensate for constant respiratory losses. This means that any classification scheme capable of identifying key differences in productivity and metabolism, will also be a useful predictor of key elements of the system's DO status.

The productivity of all aquatic ecosystems is unequivocally reliant upon nutrient availability; however, as explained in Section 2.5, nutrient concentrations are not a reliable indicator of trophic status in this region because many waterbodies experience complex natural seasonal successions that constantly alter the ecosystem's productivity base. Over the course of the hydrological year it is possible for an individual site to pass through sequential states where productivity is alternately dominated by limnetic heterotrophy, light-limited limnetic autotrophy, nutrient-limited limnetic autotrophy, light-limited benthic autotrophy and nutrient-limited benthic autotrophy. As a result the relationship between primary productivity and the concentrations of nutrients contained within the water column can also vary enormously over time.

Notably, most water quality studies are premised on the notion that there is a direct correlation between productivity and nutrient concentration, but this is only a valid assumption if waters are in a state of nutrient-limited limnetic autotrophy – i.e. if productivity is dominated by phytoplankton and is not light-limited. In fact some of the aforementioned states can result in the development of inverse correlations. For example, in waters dominated by benthic autotrophs, very low nutrient concentrations can be symptomatic of very high productivity. This is because, unlike phytoplankton, which ultimately just convert dissolved nutrients into suspended nutrients, benthic algae and macrophytes actually remove nutrients from the water column.

The spatio-temporal distributions of nutrients within a waterbody are also influenced by many of the same limnological factors that drive DO fluctuations, so it is not always as easy as is often assumed to collect representative samples, especially when dealing with sites that exhibit strong vertical stratification and/or which contain patchy assemblages of dense macrophyte stands. Accordingly nutrient data obtained by employing conventional periodic near-surface grab sampling techniques (and most have been) are often only indicative of localised conditions in the immediate proximity of the sampling point and do not necessarily provide any realistic estimate of the waterbody's active nutrient pool.

In order to redress these problems it will be necessary to develop methods of identifying and dealing with inherent differences in the ways that nutrients are processed and partitioned at different times and places within regional river systems. As is the case with DO, a classification scheme capable of recognising inherent differences in the trophic dynamics of different waterbodies would be a very useful first step towards the achievement of this objective.

The dynamics in question are strongly underpinned by limnological processes that are governed by localised physical factors such as depth, flow, wind and substrate type, and there is an urgent need to take steps to ensure that information about these variables is incorporated into ambient water quality databases. However, it is the availability of light within the water column (i.e. underwater light climate) that ultimately determines the spatio-temporal distribution of primary production within a waterbody, and it is this issue that became the primary focus of the current study.

There are some streams that receive sufficient shade from riparian vegetation to significantly affect light climate. However, tropical sunlight is so bright that the water column only needs to be directly illuminated for a short time, especially during the middle of the day, in order to support significant autotrophy. Hence shade is only a major factor for small narrow rainforest streams with dense closed riparian canopies, and there are very few such streams in the TRIAP study area (which does not include the Wet Tropics).

Accordingly the underwater light climates of most riverine waterbodies in this region are mainly dependent on water transparency.

Section 4 discusses the different methods that can be used to estimate water transparency in monitoring programs and assesses the potential for this parameter to be employed in classifications. Water transparency can be measured either by taking light attenuation readings or by employing less expensive indirect methods such as Sechhi disc measurements. Unfortunately very few monitoring programs in this region have used any of these methods and the datasets available to this study contain virtually no actual water transparency data. Transparency must instead be inferred from putative correlates such as turbidity, SPM and colour, and for a variety of reasons, this is a highly imprecise and unreliable procedure that yields extremely coarse estimates at best. Most existing datasets contain sufficient turbidity and/or SPM data to be able to gain some basic insights into the ambient water clarity characteristics of regional waters, but replication is inadequate to properly resolve hydrographic, seasonal and/or interannual variability patterns.

Representation of the critical late dry to pre-wet season period is particularly poor, as is the case for all other water quality parameters. This is a problem that urgently needs to be addressed if water quality monitoring is ever to become a truly effective tool for assessing ecosystem health in this region, because drought resistance is such a crucial determinant of ecological outcomes that it is impossible to evaluate the condition of an ecosystem without checking its capacity to cope with seasonal and interannual dry spells. In order to accomplish this it imperative to monitor the condition of drought refugia during the potentially very stressful months leading up to the first major flow event of the wet season.

Data relating to water colour are extremely limited and are absent from most source datasets. This is not a major deficiency for the many systems where transparency is mainly influenced by inorganic turbidity, but it can be a significant problem when dealing with productive wetland systems and/or the river reaches they drain into, because they can produce sufficient colour-forming dissolved organic matter to substantially reduce water transparency.

Section 4 proposes some very coarse turbidity classifications that would allow water quality datasets to be used to identify extreme contrasts in light climate. Seasonal and interannual variations in light climate class could then theoretically be determined and used to devise site classifications indicative of contrasting light climate regimes. However, even if all of the existing data limitations were to be rectified, and there are no prospects of this happening in the foreseeable future, this approach has one severe intrinsic limitation that cannot be overcome; light availability is just as dependent on the depth of the water column as it is on water transparency. Accordingly, unless the water is so opaque that it virtually prevents light from entering the water column or so transparent that it would admit light the bottom of any local waterbody, it is impossible to interpret the ecological significance of ambient water transparency values without taking bathymetry into consideration.

For this reason Section 5 proposes the use of a derivative parameter, termed optical depth, which integrates the interactions between water depth and water transparency. It is defined as the ratio of the water depth to the depth of light penetration. In most routine field-based applications the depth of light penetration would be estimated by determining the total water depth (Z) and the euphotic depth ($Z_{eu}$), and optical depth would be calculated as Z: $Z_{eu}$. However, for specialised research and monitoring applications it may be desirable and/or necessary to derive optical depth estimates from different parameters applications. For example if conducting detailed studies of macrophyte dominated wetlands it may be feasible to use the depth of the submergent plant canopy rather than the total depth.

Unfortunately there are virtually no bathymetry data available for regional river systems even though water depth plays just as important a role as transparency in determining how much of the water column and/or the benthos lies within the photic zone. In fact the lack of bathymetry data is arguably the greatest existing impediment to the interpretation of monitoring data and understanding of regional freshwater ecosystems, because depth is a key driver of virtually all of the processes that govern habitat quality and availability; such as mixing and stratification, evapo-concentration, standing water volume and water residence time.

Accordingly collection of bathymetry data is considered imperative for all future freshwater monitoring and assessment programs. At a minimum, representative depth profiles should be obtained at all active ambient monitoring sites, and the streambeds of priority sites should be surveyed in sufficient detail to be able to obtain reasonable estimates of standing water volume. This would be a significant undertaking but it is not unachievable because there are many bedrock constrained systems that would only need to be surveyed once, and the geomorphology of most systems is sufficiently stable to ensure that surveys would only need to be repeated occasionally, for example after very large, low recurrence interval floods. Once survey data are available bathymetry can be routinely monitored by simply measuring the water depth at an established reference point each time the site is sampled. The resulting data have numerous potential applications, but the most important in the context of the current study is that they can be used to assess the optical depth status of entire waterbodies.

Section 5 explains some of the ways in which this information can be used. For example each time monitoring is conducted at a site it will be possible to predict how much of a waterbody's benthos, in areal percentage terms, was photic and therefore capable of supporting autotrophy. Similarly it will be possible to predict how much of the water column is autotrophic. By comparing the ways in which these values change over time at different sites it will be possible to identify fundamental differences in the productivity regimes of different waterbodies, and this provides a sound ecological basis not only for the development of a heuristic classification scheme but also for risk assessments. For instance, optical depth and productivity regime are valuable indicators of a site's susceptibility to acute impacts from disturbances such as those caused by pre-flush rain events or the instream activities of livestock, both of which can cause sudden increases in optical depth, resulting in impaired photosynthetic oxygen production and ultimately, severe episodes of hypoxia.

Section 5.2 points out that significant changes in optical depth are often so plainly visible to the eye that rapid qualitative assessments can be carried out using simple visual observation techniques or, more importantly, from conventional photographic images. This section proposes that photography can be a valuable adjunct to conventional monitoring techniques, and illustrates by example, the kinds of valuable information that can be obtained if photographic monitoring is strategically planned. In order for monitoring programs of this sort to be successful they must be designed and supervised by professionals experienced enough to be able to properly interpret images. However, since the field work requires no special equipment and can be carried out by virtually anybody with minimal training, there is great potential for the collaborative participation of community-based monitoring groups.

All of the monitoring parameters and methods discussed thus far can be a valuable aid for assessing and classifying individual sites and/or waterbodies. However, since many key aspects of the water quality at an individual site are not only controlled by intrinsic limnological and hydrogeomorphic characteristics, but can also be strongly influenced by extremely localised factors, it can be extremely difficult to extrapolate site-based findings to whole-of-river-reach scales. Basically the proposed classification methods provide a meaningful basis for assessing and comparing sites but they offer few if any insights into the status of the large sections of river that lie between them – in fact the data available for intervening reaches are often so limited that it can be difficult to ascertain if there is water present, let alone predict its quality. In order to overcome this problem it will be necessary to gain a broad overview of entire drainage systems and this will only be accomplished by employing remote sensing techniques.

Section 6 discusses the potential for Landsat imagery to be used to obtain, not only valuable information about the location, size, permanency and morphology of most of the waterbodies contained within the river system, but also useful insights into their optical depth characteristics. Landsat senses the spectral characteristics of the sunlight reflected by bodies of water. If the water is sufficiently transparent and/or shallow to allow the sensors to see the light reflected by the substratum, the waterbody is said to be optically shallow. Conversely, if the water is too deep and/or opaque for the sensors to see the bottom it is said to be optically deep. This differentiation is only possible if the water and the substratum have contrasting spectral reflectance properties, and in most rivers they do – streambeds being composed of sand, gravel and rock, while the suspensoids in the water column comprise mainly clays, colloids and plankton.

There are of course some waterbodies, especially in off-channel wetland systems, where the spectral contrasts between the water and the substratum are weaker and more difficult to detect. This could be the case for example, if the substratum is muddy and the water column becomes turbid due to resuspension of bottom sediments. More significantly, the optical properties of waters and substrata vary substantially over time and space both within and between rivers, so extensive field calibrations will be needed in order to successfully exploit this method.

The optical depths determined from satellite images are analogous to those that would be measured in ecological assessments, but they are not the same. From an ecological perspective water is optically shallow if it simply allows light to reach the bottom – it does not have to be sufficiently transparent or shallow to allow the light reflected from the bottom to pass through the water column a second time in order to be seen at the surface. There are many waters in this region that would be appear to be optically deep in satellite images because they are too turbid or deep to enable detection of bottom reflectance even though they are optically shallow enough to maintain photic conditions within their benthos. It is possible that discrepancies such as this will eventually be overcome by developing more sophisticated methods of analysing remote sensing data, but that would require extensive research to more closely examine the relationships between spectral reflectance, optical depth and water clarity. Nevertheless, regardless of whether or not this is accomplished, the relatively coarse semi-quantitative estimates of optical depth that can already be obtained from satellite imagery are still very useful and currently represent the only viable means of obtaining any realistic indications of how light climates vary throughout entire river systems.

Discrete Landsat images are cost-prohibitive for most environmental monitoring applications, but composite images which are broadly indicative of seasonal conditions are freely available. Since this imagery dates back to 1972, it can be used to retrospectively assess historical interannual and seasonal variability patterns. One of the recurrent themes of this report is that the ecological significance of any given water quality condition can vary enormously depending on when it occurred and what the antecedent conditions were like. Basically, in this region of Australia, average and/or instantaneous water quality conditions will never provide an ecologically meaningful basis for assessing or classifying inland waters until the water quality regimes of the rivers are understood. In other words it is imperative to find out how the quality of the water contained within river systems varies over space and time, and analysis of pre-existing satellite imagery provides a unique opportunity to rapidly advance our knowledge in this regard.

# 2. NATURAL DRIVERS OF WATER QUALITY

## 2.1 Rainfall and hydrology

Availability of water is undoubtedly the primary factor governing both the water quality and the ecology of freshwater ecosystems in the northern region. The climate of the entire TRIAP study area is characterised by winter drought (see for example Figure 1). Rainfall occurs mainly during summer months reaching a peak in January or February, and as a consequence the climate is often described as being wet-dry. However, as can be seen in Figure 2, summer rainfall varies more than 5-fold across the region, and there are large catchment areas that might be more accurately termed dry or seasonally dry.

**Figure 1: Average monthly rainfall for August**

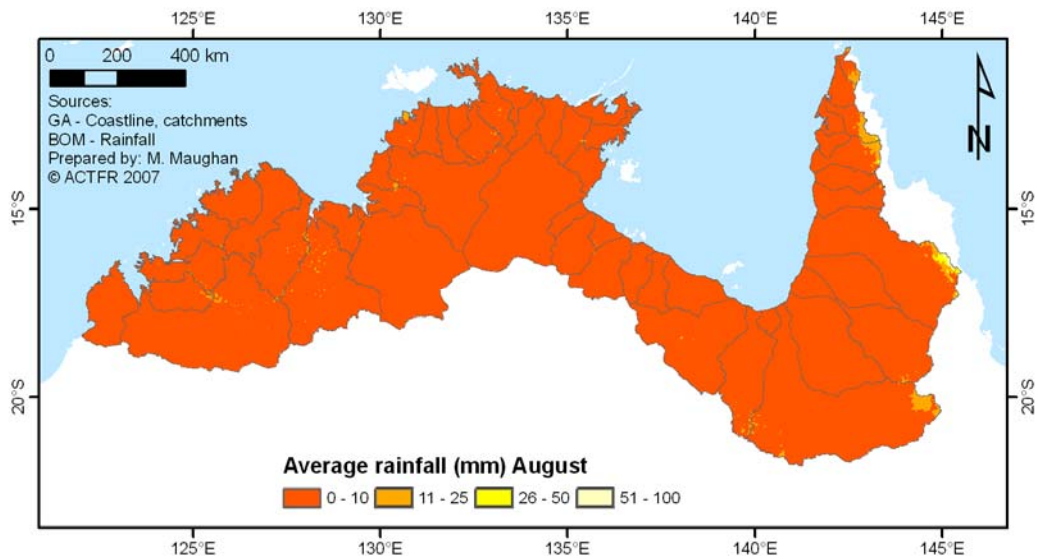
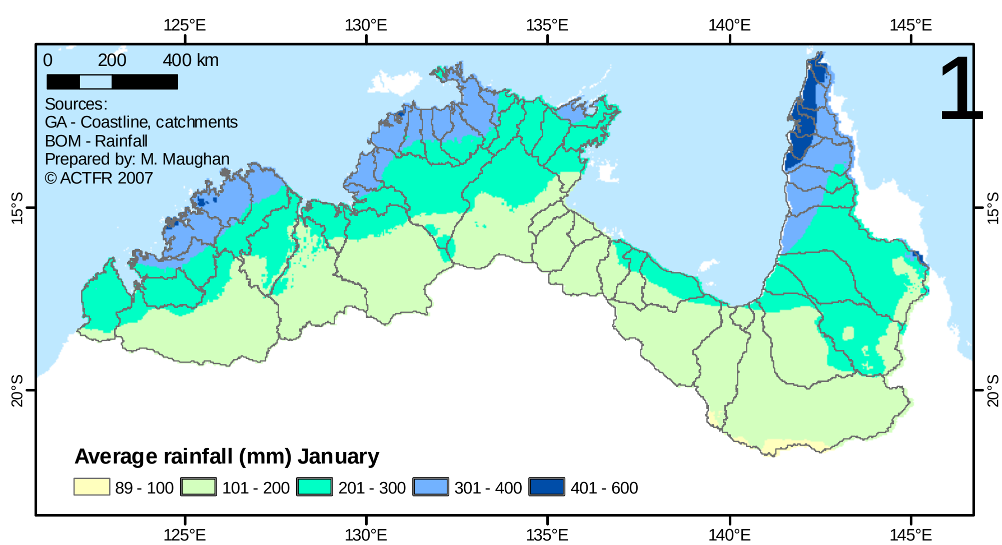


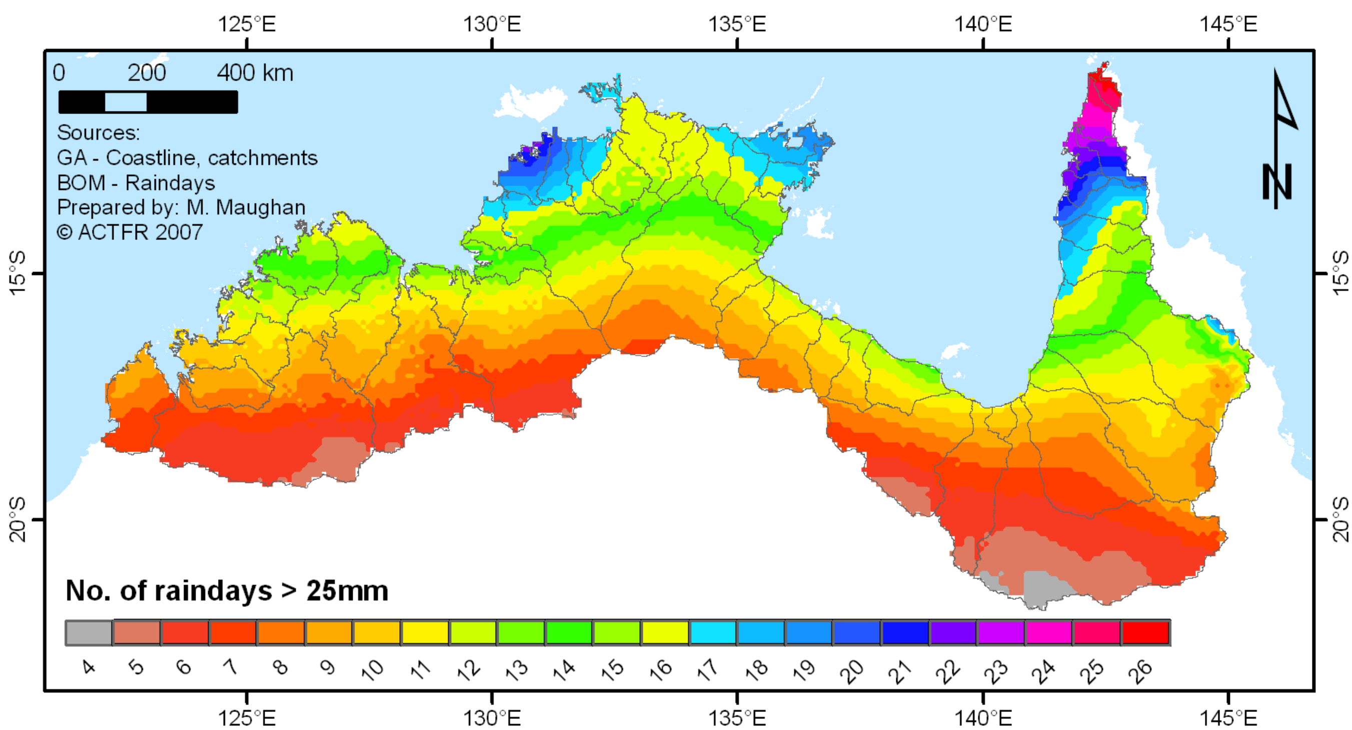

**Figure 2: Average monthly rainfall for January**



As Figure 2 shows, the wettest catchments are all located in close proximity to sections of the northern coastline that have a north-westerly aspect. Monthly precipitation declines significantly with increasing distance from the coast, generally following a south-easterly gradient. In the wetter coastal areas the winter drought usually lasts about 3 to 4 months, with some significant rainfalls being registered as early as October/November, but in the drier inland catchments the annual drought can last more than 9 months, and interannual droughts are quite common.

In the coastal regions the first post-drought rains are typically light showers and/or isolated storms that generate very little runoff, so as far as the rivers are concerned the drought does not really end until the more intense summer rains arrive. Even then it may take considerable time for the catchment soils, aquifers and watercourses to become sufficiently saturated for significant flow to occur in the rivers. Ostensibly in these higher rainfall areas there is often a distinct pre-wet season during which the catchments can become green even though stream flows are minimal. This is not nearly so evident in the drier inland catchments. As can be seen in Figure 3 these catchments often receive so few significant rain events that no clear intra-seasonal pattern can emerge – the wet season is not actually the time of the year when it rains, but rather the time when rainfall will occur if it is going to. In this case stream flow is often extremely episodic and is driven by runoff from a few brief but intense deluges, mainly associated with tropical low pressure systems. When these events arrive the landscape is often quite dry and has poor groundcover, so catchment erosion rates can be high.

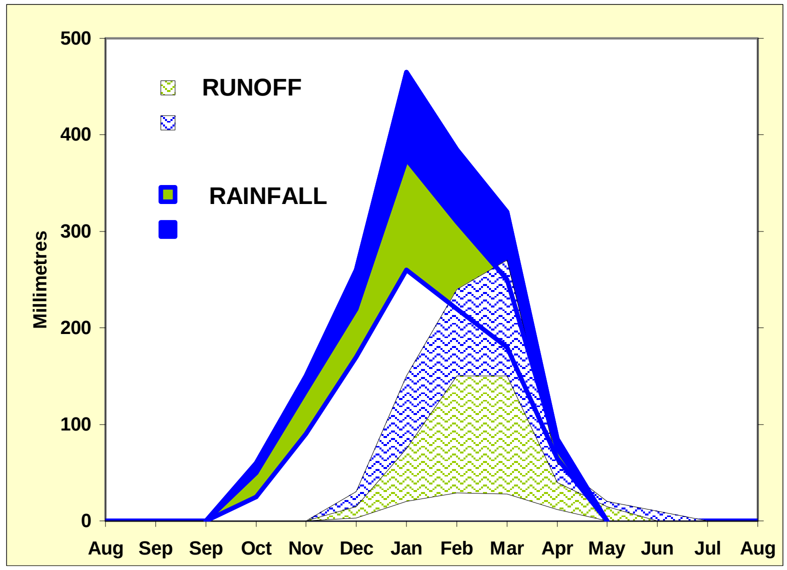
**Figure 3: Mean number of rain days registering more than 25 mm**



Due to their flat topography and permeable land surfaces, the wetter watersheds in the northern coastal regions of the Northern Territory and Cape York Peninsula, have a particularly high capacity to detain early wet season rainfall.

Accordingly, as can be seen in Figure 4, many rivers don't receive sufficient runoff to generate significant stream flow until at least November-December (Kingston 1991). This effect is particularly pronounced in low-lying catchments that contain extensive wetlands and/or groundwater aquifers. The flows in rivers receiving runoff from such catchments generally build over the remainder of the wet season and may continue to rise late in the season even after rainfall rates have begun to decline. However, the statistical picture portrayed in Figure 4 can be a little misleading because once the catchment and river channels have been fully wetted and elevated wet season baseflows are established, the system becomes so responsive to subsequent rain events that real-time wet season hydrographs are often highly erratic, as will be seen in later figures.

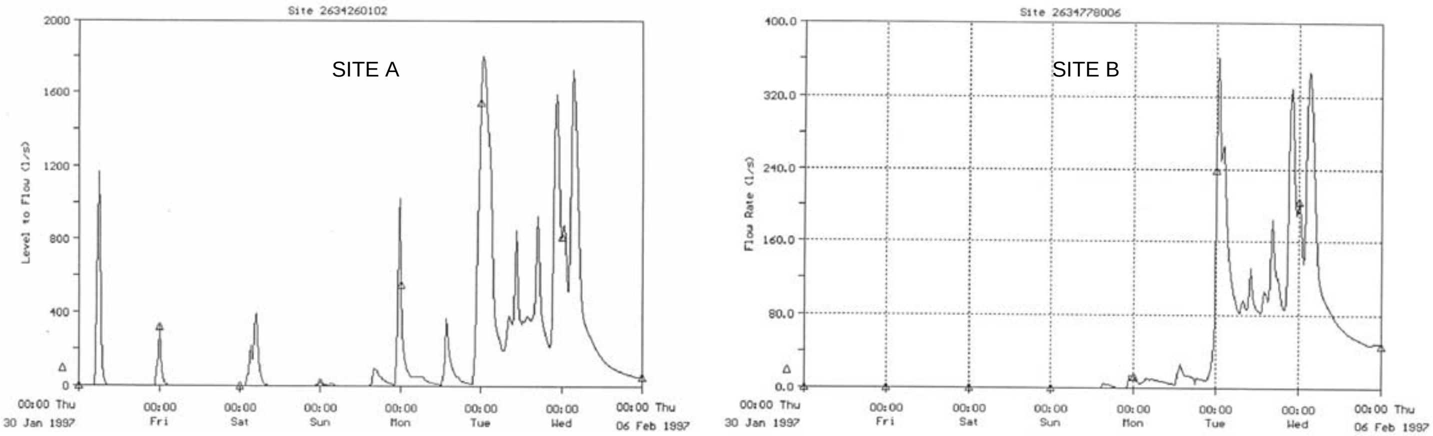
**Figure 4: Seasonal trends in rainfall and stream flow in the Northern Territory's Adelaide River basin (Adapted from Kingston 1991. Rainfall based on Darwin. Flow regionalized for 6 stations using 30 to 70 % occurrence probabilities.)**




In areas such as the Kimberleys there are catchments dominated by impermeable land surfaces and which would therefore be expected to deliver runoff to the river much more directly. In fact it is likely that many river basins would contain some streams that drain subcatchments of this kind, and although in some cases these may feed into coastal wetland systems rather than directly into the river, they are still hydrologically distinctive components of the river system and would ideally be taken into consideration in any classification scheme. Streams of this kind often flow whenever there is rain somewhere in the catchment and may cease flowing very soon after. Since wet season rains typically comprise brief intense downpours rather than sustained showers, the hydrographs of these streams can be exceptionally erratic.

It is pertinent to note that catchments can comprise a complex mix of subcatchment types and in some cases this heterogeneity can be expressed at extremely fine spatial scales. For example a paired-catchment study carried out on two small third-order streams located near Townsville (ACTFR unpublished), showed that there were substantial differences in the runoff characteristics of the two catchments, even though they were immediately adjacent to one another and appeared to be of quite similar landform. The hydrographs obtained at these two sites during the beginning of the 1996-1997 wet season are compared in Figure 5. Note that both catchments had been extremely dry until January 1, 1997 at which time each experienced minor rainfall of approximately 25 mm. Site B did not flow on that occasion but Site A flowed for 3 hours.

**Figure 5: The hydrographs of two streams that drain very small (ca. 1 km$^2$) adjacent subcatchments with apparently similar landscape features**




Site A exhibited strong flow during each subsequent rain event and in most cases stopped flowing again within a few hours after cessation of rain. Weak baseflows were maintained for about a day following very heavy rain (totaling 600 mm) on the 4th and 5th of February, and for almost 2 weeks in the aftermath of a cyclonic flood event on the 23rd of March (not shown). Nevertheless, in all other respects the hydrograph of site A was so closely correlated to precipitation that for the purposes of this discussion it was not worth including a separate rainfall plot. This stream retracted to a few lentic pools during April and dried out completely by May. In contrast site B did not begin to flow until its catchment had received substantially more than 200 mm of rainfall, but it did not stop flowing again for at least 6 months (at which time monitoring ceased). Once baseflow was established at site B the stream became just as responsive to rain events as site A and the striking similarities evident in the event hydrographs of February 4th and 5th were maintained for the remainder of the wet season.

The ecological consequences of the observed differences in baseflow duration were quite striking. Site A remained turbid until it had virtually dried out and never developed any substantial faunal or floral community. Site B on the other hand rapidly ran clear between spates and was quickly colonised by benthic algae, a diverse range of macroinvertebrates and eventually three species of aquatic macrophyte.

Rainfall is unevenly distributed across larger catchments so it is seldom feasible to make the simple direct comparison that was possible in the above example. Nevertheless, the hydrographic diversity observed in these smaller streams is still discernible in larger systems, and it would be unrealistic to expect adjacent tributary systems or consecutive reaches of a large river to necessarily exhibit the same hydrology. The hydrographs of large streams are in fact surprisingly similar to those of smaller systems except that event durations increase in proportion to catchment size, with spates generally lasting for hours to days in small streams, and days to weeks in large rivers.

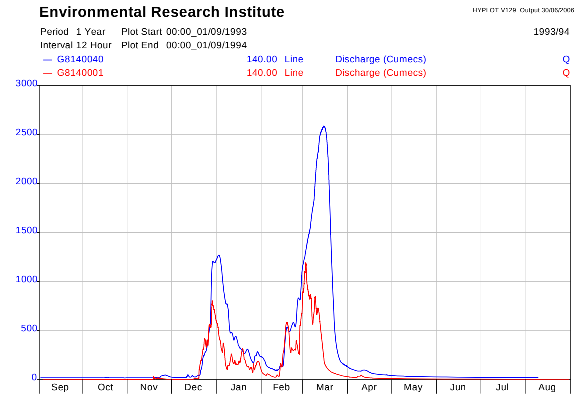
The gradual seasonal build-up in stream discharge portrayed for the Adelaide River in Figure 4 is at least partially evident in the real-time hydrographs of several Northern Territory rivers (Moliere 2008). The Daly River catchment for example would be expected to behave in this way since it contains an extensive karst aquifer system that detains sufficient runoff to be able to maintain strong baseflows throughout the dry season. The hydrographs in Figure 6 confirm that as expected, significant storm flows do not occur until December. However, the anticipated build-up in discharge rates over the latter half of the wet season is only really obvious in very wet years – in normal years rainfall is simply too episodic for a clear overall flow pattern to emerge. Basically, in any given wet season, water level changes and flow peaks intensive enough to substantially disturb instream habitats and/or flush out the river system can occur any time after November. Notably, however, baseflows generally build over the course wet season, and even in the driest years it can take months for the flow to recede to typical dry season levels.

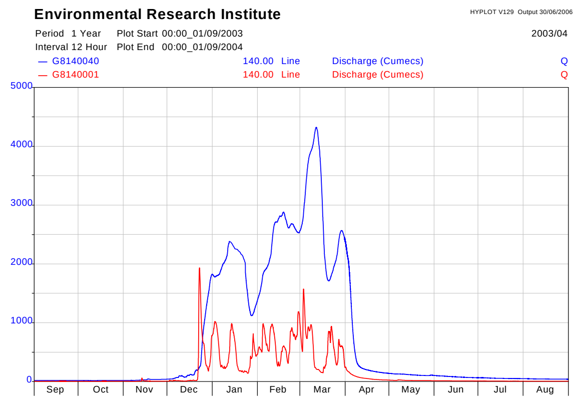
**Figure 6:** **Selected hydrographs from the upper (red) and lower (blue) Daly River in the Northern Territory**

**Top Right:** **An average rainfall year**

**Bottom Right:** **A very dry year**

**Bottom Left:** **A very wet year**



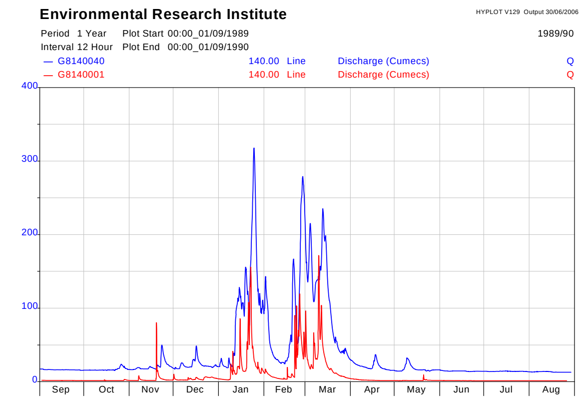
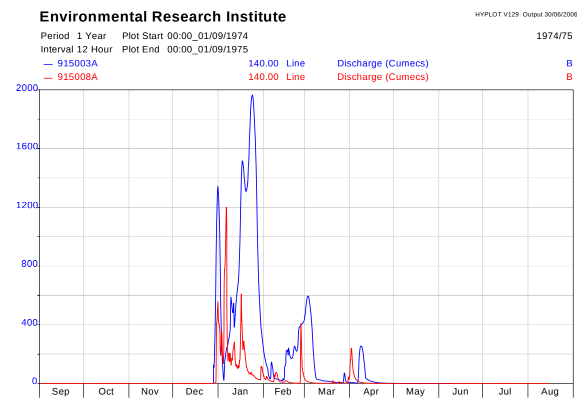



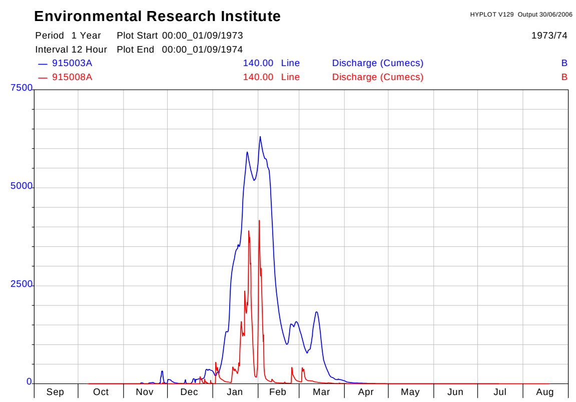

**Figure 7:** **Selected hydrographs from the upper (red) and lower (blue) Flinders River in Queensland**

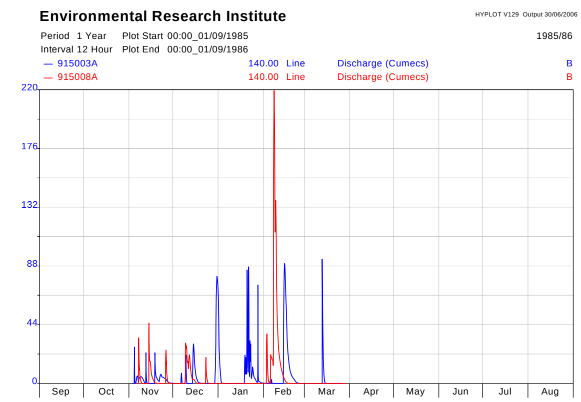
**Top Right:** **An average rainfall year**

**Bottom Right:** **A very dry year**

**Bottom Left:** **A very wet year**



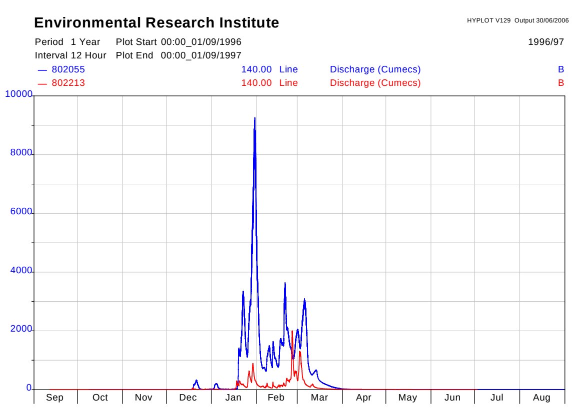



The hydrographs in Figure 6 are plotted on a linear scale, and because the scaling has been adjusted to accommodate major event flows, it is difficult to discern baseflows and minor events. (Some of the effects discussed here can be much more easily seen on log plots, but these have not been included here due to space constraints). However, peak flow rates in the "very dry year" plot are ten times lower than other years, and as a result it is possible to see that there are some very small pre-wet season flow disturbances. These indicate that some runoff does enter the river at that stage of the year. Minor stormwater inputs of this kind are too small to be of much interest to hydrologists and would not be expected to alter hydraulic habitat conditions, but they do have the potential to significantly influence water quality. This is discussed later in the report.

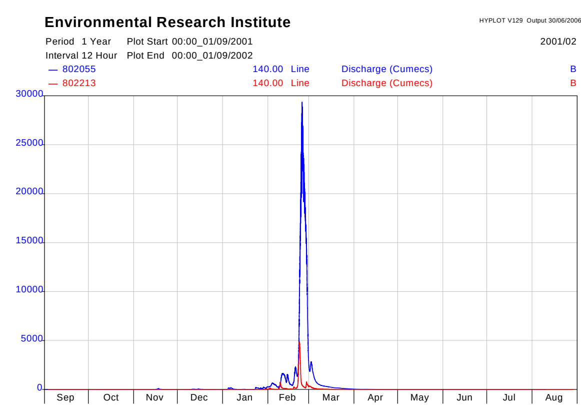
These minor pre-wet inflow events are strikingly evident in the "dry year" hydrograph of Queensland's Flinders River, shown in Figure 7. This catchment receives far fewer rain events than the Daly (see Figure 3) but in most years the Finders River actually experiences more small spates, suggesting that the catchment may have considerably less detention capacity. Regardless of the reason it is evident that at least some runoff is delivered to the river during most significant rainfall events. The pulses generated by these inflows subside rapidly, and flow rates often fall back to very low levels between events. In fact strong baseflows are maintained for no more than 4 to 5 weeks at a time during most wet seasons, and for less than 3 months even in the wettest year. Basically the wet season hydrograph of rivers like the Flinders have almost no tail, and as a consequence, flows are very sluggish or absent most of the time, even during the wet season.

**Figure 8: Selected hydrographs from the upper (red) and lower (blue) Fitzroy River in Western Australia**

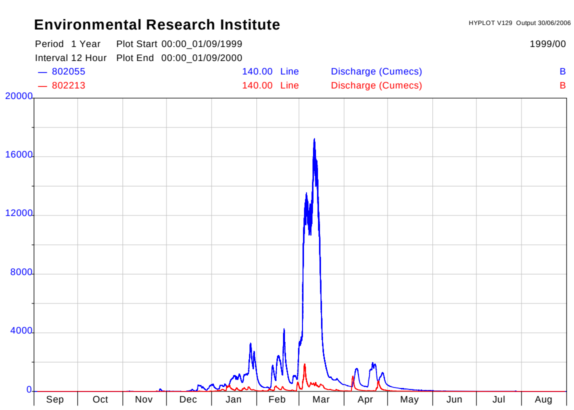
### A) An average rainfall year




### B) A Major flood year




### C) A very wet year




### D) A very dry year

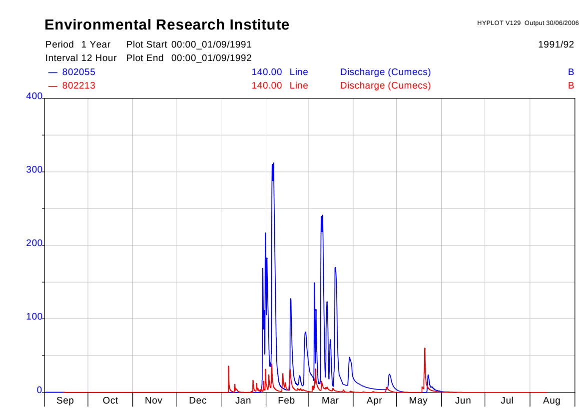


Figure 8 suggests that the lower reach of the Fitzroy River in Western Australia behaves quite similarly the Daly River. The conceptual pattern portrayed earlier for the Adelaide River is more strongly evident in these hydrographs, with flow events rarely being registered before mid-January and baseflows generally building to a maximum during February-March. However, the tail of the hydrograph is steeper and less pronounced than it is in the Daly, meaning that baseflows recede more rapidly and to lower levels. The hydrographs for the upper reaches of the Fitzroy River more closely resemble the Flinders in that there is little evidence of the establishment of strong wet season baseflow, events being manifested mainly as pulses with little or no tail. Notably though there are far fewer small spates and no clear indication of pre-wet events suggesting that the upper Fitzroy has higher detention capacity than the Flinders. Given the heterogeneity of natural catchments it is likely that there would be at least some tributary systems that resemble the Flinders even more closely, but there are insufficient gauging data to fully test this contention.

The elevated baseflows observed in some systems during and/or soon after the wet season may be supplemented by surface inflows of water from adjacent wetlands (this can be return flow of floodwater that originated from the river and/or outflow of water collected from the wetland's own catchment). However, baseflows that persist during the dry season can be attributed solely to groundwater inputs. Early in the dry when regional water tables are still elevated these can arise from numerous widespread sources, and especially the alluvial aquifers associated with the river. However, any baseflows that are present at the end of the dry season usually originate from just a few discrete aquifers that only feed a few selected tributaries and/or river reaches.

It must be stressed that gauging stations only record the surface flow that is present at a particular point in the drainage system. Many rivers contain deep basal sands and these can support significant hyporheic flow for a long time after gauging stations stop registering. This cryptic subsurface flow can play an important role in maintaining and/or flushing instream waterholes. Nevertheless, the majority of streams still eventually stop flowing entirely during a normal dry season, and as a consequence they either dry out or retract into a series of lentic pools. Permanent pools can form around groundwater springs or in places where the streambed either intercepts a water table or is perched on a substratum that is sufficiently impermeable to prevent seepage losses. In the latter case the perched streambed can still contain extensive basal sand deposits so the visible surface water may represent only a small proportion of the waterhole's true volume. Permanent waterholes that are not supplemented by subsurface water reserves must be deep enough to resist evaporation (discussed later). The size and number of pools can vary enormously between streams, and since their formation depends on very localised and often anomalous geomorphological features, their existence can be extremely difficult to predict.

### 2.1.1 Water Quality Implications

Though hydrology is undoubtedly the primary driver of water quality processes, hydrographic records alone do not provide a basis for assessing or predicting water quality and/or instream habitat conditions. Notably stream gauging records provide no indication of whether water even remains present after flows stop registering (or in fact if there is still significant hyporheic flow in the stream). Ostensibly there is no way of telling if there is an aquatic ecosystem present, let alone what its condition might be. Moreover, it is not flow alone, but rather the interactions between flow and the receiving environment that determine water quality outcomes. For example the size of the receiving water body is a critical factor – for example very minor baseflows that would have negligible effects on a large body of water can be sufficient to constantly flush a small waterhole.

The ionic composition of river water depends primarily on the water's source. During the wet season water enters streams from many different sources and in different relative proportions at different stages of the hydrograph, so ionic composition can potentially fluctuate quite rapidly (although in many of the wetter northern catchments ionic strengths may simply remain very low all of the time) . But as discussed previously, the water contained in most rivers later in the dry season originates from just a few specific sources, so for much of the year ionic compositions vary much more gradually and predictably.

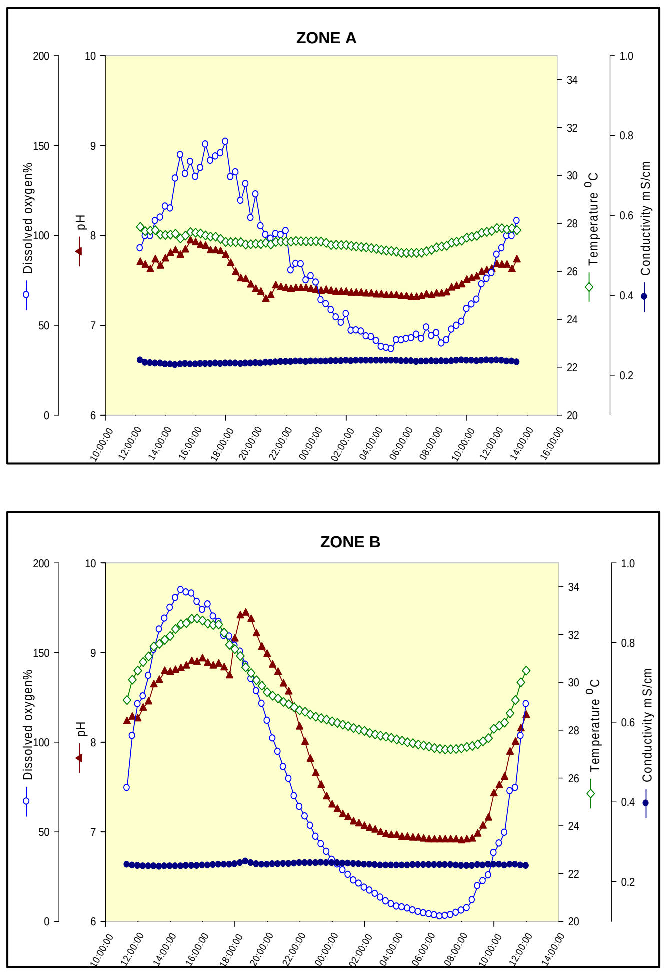
The major ionic constituents of local waters are not subject to substantial biological alteration, so even though evaporation may cause concentrations to increase over the course of a dry season, the fundamental ionic signature of the water can still be retained. Most other water quality parameters, especially dissolved oxygen (DO), carbon dioxide (and therefore pH), nutrients, colour, temperature and chlorophyll do not behave in this way. Under the limnological conditions that typically develop over the dry season these parameters are so strongly influenced by natural biological processes that the aquatic ecosystem itself is very often the major determinant of the concentrations in the water column.

## 2.2 Limnology

Most streams stop flowing at some stage during a normal year and even the few that don't generally flow so slowly for most of the year that they exhibit limnological traits more characteristic of lakes than streams. However, they differ from normal lakes in several important respects. For instance they are long and thin, and are normally sheltered by high banks and/or riparian vegetation. Consequently they often have minimal wind fetch and very low mixing and re-aeration capacity. They are also often shallow enough to enable most of the water column to maintain fairly intimate contact with the very high concentrations of biota and nutrients contained within the benthos. This, linked with consistently high temperatures, means that they are often highly productive and can have extremely high respiratory oxygen demand.

These waters can seldom exchange gases with the overlying air efficiently enough to compensate for the effects of biological consumption and production processes. All of the aerobic organisms in a water body constantly respire, consuming dissolved oxygen (DO) and producing carbon dioxide in the process. However, during daylight hours submerged autotrophs carry out photosynthesis, and in the process they assimilate carbon dioxide and produce surplus DO. Since photosynthetic activity is driven by solar radiation DO concentrations exhibit a diel periodicity that recapitulates the variations in sunlight intensity over the course of each day. During a typical fine dry season day the resulting pattern can be remarkably predictable, as shown in the examples in Figure 9. Biological DO production is so sensitive to light intensity that passing clouds can introduce considerable random variations to this pattern, but that is not usually a major issue during the dry season.

**Figure 9: Variations in DO, pH and temperature observed over the course of a single day at two locations in the near-surface layer of one small waterhole in the Burdekin River.**




The fluctuations shown in this figure are quite normal for regional waters, in fact far larger temporal and spatial variations are regularly encountered, especially towards the end of the dry season. Note that the monitoring stations represented in the two plots were located only about 20 m apart, yet the differences between them are very ecologically significant. For example in the upper plot DO never fell below 35% (a level that would not be expected to acutely affect even the most sensitive species) while in the lower plot DO fell to levels low enough to rapidly asphyxiate many fish and macroinvertebrate species (Butler and Burrows 2007a, b).

The plots in figure 9 show that pH values can also fluctuate substantially over the course of a day. This is because free carbon dioxide dissolves in water to form carbonic acid, meaning that pH values become less acidic when carbon dioxide is consumed and more acidic when it is produced. Eventually pH levels become so alkaline that carbonic acid and free carbon dioxide can no longer exist in the water. Some autotrophic species stop photosynthesising under these conditions but others, including cyanobacteria and many macrophytes, are able to enzymatically extract carbon dioxide from bicarbonate ions. This process yields hydroxyl ions as a by-product and as a result pH can rise to very high levels (in excess of 10.5 in extreme cases). Most fresh waters in this region have only low to moderate alkalinity, so they have limited capacity to chemically resist these pH fluctuations. Nevertheless, pH participates in so many chemical reactions that diel periodicity is not as predictable as it is for DO.

Given the severity of typical overnight oxygen sags, episodes of mortality are surprisingly infrequent. This is because the fauna that occupy habitats of this kind are so well attuned to these natural daily spatio-temporal fluctuations. For example most of the very hypoxia-sensitive species are mobile and are very adept at being in the right place at the right time to obtain DO, while many of the less sensitive species have physiological adaptations that allow them to survive brief periods of exposure to quite severe hypoxia. However, there are also large numbers of sedentary species that simply avoid microhabitats that do not contain sufficient DO to meet their needs. Accordingly DO variations of this kind are an important natural determinant of habitat utilisation and secondary productivity.

The biological oxygenation system is so delicately balanced that severe life-threatening episodes of hypoxia can develop rapidly if photosynthetic production is impaired during daylight hours and/or if there is a sudden increase in respiratory consumption rates. In small lentic waterbodies, problems of this kind can be brought about by fairly minor localised disturbances. For example, by turbating bottom sediments and/or depositing excrement, wading animals can increase the water's turbidity and/or oxygen demand sufficiently to cause a major DO sag.

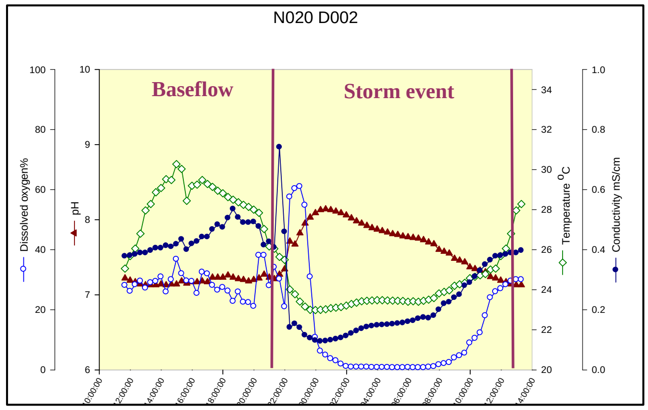
Many waters are only transparent enough to support benthic photosynthesis when strong direct sunlight is available so they can become quite hypoxic if weather conditions become overcast. This can create a particularly hazardous situation for instream fauna because it means that conditions can become extremely stressful during the onset of storm events – and it is storm events that present the greatest hypoxia risks to most local waters. The sudden increases in water depth and turbidity associated with influxes of stormwater can severely inhibit benthic photosynthetic production, allowing oxygen deficits to rapidly develop. This is often exacerbated by increased respiratory demand due to microbial decomposition of fresh organic matter carried into the stream during the storm.

Large scale wet season storm events are not usually associated with severe or prolonged DO sags because they provide sufficient stream flow to increase mixing and re-aeration, and/or to flush away much of the oxygen demanding biomass. However, smaller pre-flush events carry sufficient runoff into streams to substantially alter their clarity and depth, without generating any sustained stream flow (as can be seen in the hydrographs in section 2.1). The resulting episodes of acute hypoxia are the most common and widespread natural source of fish kills in northern Australia, and in many instances a single isolated event of this sort can be the ultimate determinant of a waterbody's net annual contribution to regional fish productivity.

Note that the quality of pre-flush runoff can be very poor and many other factors such as natural toxins have also been implicated as potential causes of fish mortality. However, none of these are as ubiquitous as hypoxia episodes, and in most reported cases hypoxia is still likely to have been a potential contributing factor anyway. This is because fish increase their gill ventilation rates when exposed to hypoxic water quality conditions, and this concomitantly increases the rate at which toxicants can be taken up through the gills. Recent research has demonstrated that in some species ventilation volume can increase by as much as 20 to 30-fold if DO concentrations fall to threatening levels (Butler and Burrows 2007a, Pearson et al 2003b)

The example in Figure 10 (Loong *et al* 2005) shows how DO, pH, temperature and conductivity levels changed in the near-surface water layer of an instream waterhole during a small 15 hour flow event. Stable baseflow was being maintained prior to the event and conductivity levels were relatively constant at around 0.4 mS $cm^{-1}$ (400 µS $cm^{-1}$). The arrival of storm flow is marked by the sudden rise and then dramatic fall in conductivity, and an associated increase in pH. (If the event had been larger pH would have fallen to a value of less than 6 – approaching that of rainwater runoff – but in this case the amount of runoff was too small to dilute the alkaline water carried down from upstream reaches). At the end of the event conductivity and pH both returned to normal baseflow levels.

**Figure 10: Water quality variations during a minor pre-wet season flow event in a waterhole located in the baseflow channel of a major tributary of the Burdekin River**




Note that due to overcast conditions and moderately elevated (natural) turbidity levels, DO concentrations were already quite low (30%) prior to the event. On the rising limb of the hydrograph flow rates increased sufficiently to provide some re-aeration so DO levels initially rose. However, concentrations rapidly fell to almost zero as soon as the flow rates began to recede and they remained at hazardously low levels until the stormwater pulse had been expelled by baseflows (about 12 hours after the initial DO sag). Most oxygen dependent species can asphyxiate within minutes at these extremely low DO levels, so any organisms unable to avoid this pulse of water are likely to have perished. Note that in this example the site received sufficient baseflow to effectively expel the stormwater pulse (as evidenced by the restoration of pre-event water quality conditions). However, there are many streams that don't experience significant baseflow until late in the wet season (if at all) and stormwater from pre-wet events can remain resident in these systems for prolonged periods leading to much more persistent water quality problems.

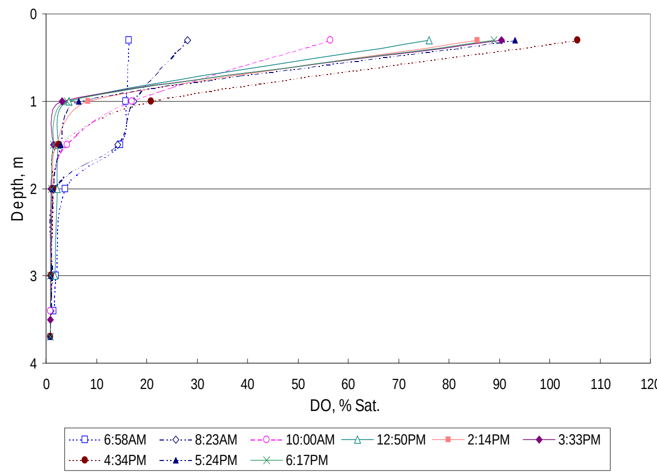
The variability patterns discussed so far relate only to readings taken from about 20 to 30 cm below the water surface. This is the part of the water column where temperatures almost always reach maximum levels. The highest DO and pH values also usually occur near the surface, especially during the types of hypoxia episodes shown in figure 10. However, under normal ambient conditions many lentic waters are so poorly mixed that DO and pH may sometimes reach their maximum levels in deeper pockets of water surrounding submerged plants.

Most local waters thermally stratify (meaning that the heat of the sun warms the surface water layer, decreasing its density to the point where it no longer mixes with the cooler waters underneath). Deep waters are often stably stratified most of the time in this region, especially during the hotter months towards the end of the dry season. Shallower waters are also prone to vertical stratification but in many cases, particularly during the cooler dry season months, the surface layer may cool sufficiently for mixing to occur overnight (this is called diurnal stratification). The bottom waters in stratified systems are prone to becoming severely hypoxic because they never get to make contact with the air and generally support low levels of photosynthetic DO production, but are subject to substantial respiratory oxygen demand from microbes living in the benthos.

Similar effects occur during summer in deep water habitats of more temperate climatic regions, and these have been widely documented in the limnology literature. However, the local situation is different in several respects. For instance temperatures are higher year round, as are respiratory oxygen consumption rates. Moreover, the density of water decreases disproportionately with increasing temperature, so the thermal differential required to prevent the water column from mixing is not as great as it is in less hot climes (near the end of the dry season daytime surface water temperatures in excess of 37ºC are not uncommon here).

The riverine waters in this region are also smaller, narrower and more sheltered and/or sinuous than the waters dealt with in most limnological publications. Accordingly they are not as readily mixed by winds.

**Figure 11: Diel changes in DO stratification in a lagoon situated in a coastal creek on the Herbert River floodplain. Total water depth was 10 metres.**



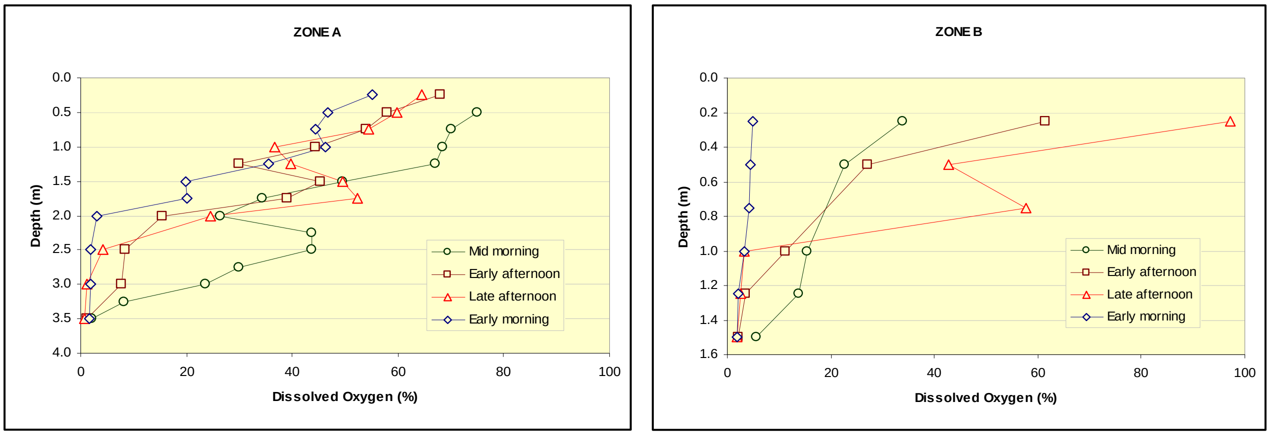

One of the major consequences of this is that even very shallow waters can stratify. The limnological literature is dominated by studies of large waterbodies with mixed surface layers tens to hundreds of metres deep. Information relating to very small warm water systems is very limited and there are few reported cases of stratification occurring in waters less than 2 m deep. In contrast our surveys of the riverine wetlands in this part of the world suggest that it is uncommon to find a lentic waterbody that does not exhibit stratification of some kind, even if maximum water depths are no more than 40 cm. Moreover, the mixed surface layer (i.e. epilimnion) of waters as deep as 10 m may be no more than 1 m deep during the heat of the day, as can be seen in Figure 11. Thermal stratification may not always be in evidence, especially in waters less than a metre deep, but average DO levels still usually decline substantially with increasing depth, regularly reaching quite low concentrations near the bottom. This can mainly be attributed to the very low mixing energy available in sheltered bodies of water and to the very high benthic oxygen consumption rates in these warm waters. pH levels can vary similarly with depth but as discussed previously, outcomes are less predictable.

Diel DO periodicity is usually evident throughout the photic zone (i.e. that part of the water column that receives sufficient light to support photosynthesis). In waters that do not support high levels of benthic autotrophy, the intensity of the diel fluctuations usually declines with increasing water depth although at certain times of the day there may be mid-water anomalies created by layers of suspended phytoplankton, especially in waters that contain high concentrations of cyanobacteria. The patterns observed in waters that support benthic DO production can be much more complex, with the intensity of diel DO excursions fluctuating wildly over space depending on the proximity of submerged macrophyte canopies and/or benthic algae beds. Macrophyte assemblages create sufficient hydraulic resistance to further reduce mixing, so the water column can become progressively more compartmentalised as plant densities increase over the course of the dry season.

The mixing depth of diurnally stratified waters fluctuates constantly over the course of the day. In cases where there is no benthic DO production, the unmixed water layer that forms near the bottom during the heat of the day can become quite hypoxic. However, at some stage of the day (or more commonly the night) this water mixes with the oxygenated surface layer and as a result DO levels near the bottom can suddenly increase. As consequence of these kinds of effects the diel periodicity patterns observed at depth may be quite different to those observed at the surface. Nevertheless average DO levels are still usually considerably lower near the bottom unless benthic productivity is exceptionally high.

The above effects can be expressed over very small spatial scales hence, as shown in Figure 12, even quite small waterbodies can contain a number of zones that exhibit distinctive temporal variability patterns (Loong et al 2005). These patterns of small scale spatio-temporal variations are usually firmly entrenched by the time flows reach normal dry season baseflow levels and generally continue to intensify over the course of the dry season.

**Figure 12: DO profiles for different times of the day at the two monitoring stations shown in figure 9.**




In the face of such substantial small scale heterogeneity it can be a major and complex undertaking to describe the status of a single waterbody, let alone an entire river reach – the established practice of attempting to assess the ambient water quality of a river reach hundreds of kilometers long by collecting discrete grab samples from a single point in the water column at time intervals in the order of weeks to months is simply not defensible.

The variability patterns shown here relate mainly to DO, pH and temperature – i.e. parameters that can be monitored intensively using dataloggers. Due to prohibitive analytical costs and logistical constraints other water quality parameters have seldom been studied intensively enough in local waters to able to quantify heterogeneity at these small scales of time and space. However, there are sound theoretical grounds to expect the observed variations in DO, pH and temperature to strongly influence not only the concentrations and chemical speciation of other parameters but also their bioavailability and ecotoxicology. Moreover, as will be discussed in a subsequent section, many of the biophysical factors that drive the limnology of local waters directly impact on the ways in which bio-active contaminants such as nutrients and trace elements are partitioned within a waterbody.

## 2.3 Timing and Sources of Inputs

The event hydrographs of virtually all streams in this region are exceptionally flashy, exhibiting sharp peaks that closely recapitulate the timing of rain events. This suggests that a significant amount of runoff reaches watercourses via rapid overland flow paths, even in catchments with high runoff detention capacity. The quality of this surface runoff varies somewhat depending on catchment type and condition. The databases accessible to this study contain insufficient data to be able to statistically quantify this variability (see appendix), but the study team have been involved in more than enough studies to be able to provide some qualitative estimates based on experience.

As a rule of thumb overland flow is acidic (pH 5.8 to 6.0), poorly buffered (has very low alkalinity) and very dilute (EC << 50 µS $cm^{-1}$) with ion ratios similar to seawater. Basically it resembles carbon dioxide saturated rainwater indicating that the runoff has insufficient time to leach much salt from the highly weathered surface soils. However, particles of dust, soil and organic detritus are mobilised and as a consequence the surface runoff that reaches low order streams almost always contains significant concentrations of suspended particulate matter (SPM). The concentration and composition of the SPM varies with rainfall intensity and catchment type, slope and condition. Catchments with land surfaces dominated by rock and those with dense groundcover usually yield only moderate quantities of SPM. Nevertheless concentrations often still reach values in the order of 50 to 150 mg/L during spates and that is substantial compared to the very low concentrations that are typically maintained in systems of this kind during periods of low flow.

Dry catchments with poorly covered erosion-prone soils generate the most turbid runoff, with SPM concentrations commonly reaching 250 to 1000 mg/L during events. (Note that these estimates relate only to materials that are fine enough to remain in suspension – primarily clays and silts – larger soil particles may also be mobilised in runoff but these do not stay in suspension long enough to be considered as SPM). Even higher concentrations are observed in these types of catchments if there is significant gully erosion. Moreover, because gullies often intercept dispersible clay sub-soils, the resulting SPM can comprise mainly colloids that are too fine to ever settle unless they flocculate (i.e. unless they coalesce to form aggregate particles that are heavy enough to settle).

The runoff collected by low order tributaries generates turbulent flows capable of eroding streambeds and banks and as a consequence, other factors being equal, turbidity levels generally increase as the waters travel through the drainage system. Streams with high bed-slope generate the highest runoff velocities and therefore have the greatest sediment entrainment capacity. However, the hydraulic head (i.e. the force that drives water flow) depends not on bed-slope but rather the gradient of the water surface, and tropical rainfall can deliver water to streams rapidly enough create significant hydraulic gradients in streams with relatively low bed-slope. Accordingly the flow velocities and channel erosion capacities of streams in this area are generally higher than they are in more temperate climatic regions.

The SPM concentrations in higher order streams and rivers often rise to levels in excess of 700 to 7000 mg/L during the rising stages of flow events, sometimes briefly exceeding 10,000 mg/l during large floods. As a general rule concentrations increase in proportion to the size of the event, but antecedents and timing are important. The pre-flush rain events referred to in earlier sections of this report often wash away most of the easily mobilised materials such as dust and humus that have accumulated in the catchment over the course of the dry season. The runoff from these events can therefore be of particularly poor quality, often containing not only elevated SPM levels but also high concentrations of organic matter, nutrients and natural toxins. These events have special significance to the river ecosystem because they occur at times when streams are at minimum flow (often stagnant) and they are generally too brief to provide much supplemental flow. Consequently the poor quality stormwater can remain in the system for prolonged periods.

The first significant flow event of the season flushes this water away carrying with it most of the easily mobilised fine particulate materials that have accumulated within the stream since the last swiftflow. This first-flush water often contains higher contaminant concentrations than the waters carried by ensuing flow events of a similar size. However, there are usually some subsequent wet season storm events that are much more intense and prolonged than the first-flush. These generate greater erosive forces leading to considerably higher concentrations of SPM (and concomitants such as particulate nutrients and trace metals). In the longer term it is the massive floods associated with cyclones and rain depressions that always yield the highest particulate contaminant concentrations.

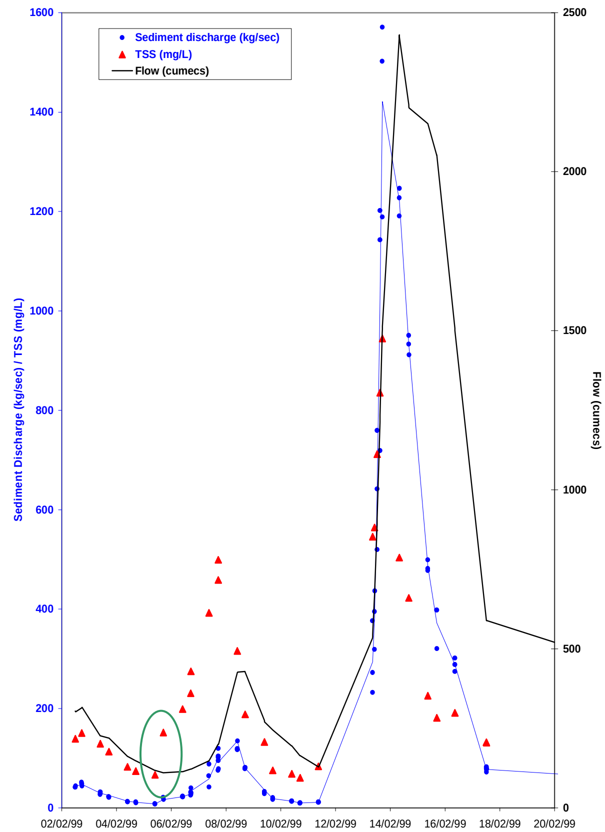
Studies of water quality dynamics have historically focused on measuring contaminant fluxes, and since pre-flush events generate little or no discharge they have received very little attention. In fact, as the data review in the appendix shows, pre-wet season conditions in general are poorly represented in available databases. However, the available database does contain some instances where monitoring was carried out during small events later in the wet season, and these provide some useful insights into the likely quality of pre-flush events.

For example the Queensland NRW data shown in Figure 13 (Butler and Burrows 2006) were collected from the Drumduff gauging station on the Palmer River during February 1999, after elevated wet season baseflows had already been established.

This figure illustrates the rapid and instantaneous increases in TSS (i.e. SPM) concentrations which inevitably take place on the rising limb of event hydrographs. Note for instance the sudden two-fold increase in SPM that occurred as soon as flow rates began to rise on 5/2/99 (points circled in green). It is also noticeable that although SPM values reach their highest peak during the largest event, the peak concentrations associated with smaller events are still substantial.

The initial rapid decreases in SPM on the peak of the hydrograph are also typical of the vast majority of regional streams. The continued decline in SPM levels during the falling limb of the hydrograph is characteristic of river systems that receive high enough volumes of true baseflow to displace most of the turbid stormwater. The falling limb is not usually as steep as this in rivers that receive inflows of surface runoff that has been temporarily detained by wetland systems. This delayed runoff generates a hydrograph with a tail similar to that created by high baseflow but, because the flow is driven by surface runoff, SPM levels can remain elevated for weeks until true baseflow is restored. There are also many streams that simply do not receive enough baseflow to fully displace stormwaters, and these may remain persistently turbid. Some streams are intermediate in this regard, receiving sufficient baseflow to be able to run clear in wet years but not in dry ones.

**Figure 13: The dynamics of total suspended solid (TSS) concentrations and sediment discharge in the Palmer River in the Mitchell catchment, Queensland.**




The preceding comments relate mainly to wet season conditions; by the late dry season most regional streams have either stopped flowing or are flowing so sluggishly that their flushing capacity is severely impaired. Accordingly the poor quality runoff generated by pre-flush events can be retained for prolonged periods, even in perennial river reaches. Moreover, since high baseflows cannot normally be restored until there has been sufficient rainfall to recharge groundwater aquifers, saturate basal sands, and in some cases, refill adjacent wetland systems, a significant number of pre-flush events may need to occur before the cumulative precipitation is adequate to accomplish this. Rivers situated in the wetter areas near the coast inevitably receive flushing rains each wet season, but this is far from guaranteed in the drought-prone inland catchments – here streams and waterholes can potentially detain pre-flush runoff for years at time.

There is a paucity of data indicative of the effects of pre-flush events or late dry season water quality in general. However, the very turbid runoff is visually obvious and has been observed on numerous occasions by field personnel and can be seen in remote sensing imagery. Satellite images for example show that some inland waterholes remain turbid for years at a time but run clear in other years; a variability pattern that is entirely consistent with the hypothesis that pre-flush runoff can sometimes be detained for years.

As implied in the preceding discussions, the groundwater inflows that drive baseflows are generally relatively free of particulate contaminants. Groundwaters are fairly well-filtered and therefore usually quite clear by the time they enter a stream. The aquifers associated with surface wetland system may sometimes contain colour-forming organic matter (COM) but most groundwaters are virtually colourless. Groundwaters may however, contain significant concentrations of oxidisable ions, especially the reduced forms of iron and manganese. If present these react with oxygenated surface waters to form strongly coloured, insoluble oxyhydroxides, leading to the development of secondary turbidity.

However, the precipitates generally flocculate and settle to the bottom quite rapidly. In some cases quite thick flocc layers can build up in certain places on the bottom, especially in the vicinity of groundwater springs, but these do not generally have persistent effects on the clarity of the overlying water column. Groundwater can also carry aluminium colloids into streams but these are usually only present in low concentrations and seldom remain in a dispersed colloidal form for long once exposed to surface water conditions.

Some groundwaters can contain ecologically significant concentrations of trace metals, and in rare cases (depending on soil type), phosphate. The flocculation processes mentioned above can play an important protective role for the ecosystem, as they can remove a large proportion of these contaminants from the water column through co-precipitation. (This is such an efficient process that laboratories sometimes add ferrous iron to water in order to prepare phosphate-free solutions).

Water levels and current velocities continually fall (albeit often very slowly) during prolonged periods of baseflow, so channel erosion is unlikely to be a significant source of dry season turbidity. However, some river reaches have sufficient fetch to experience wind turbulence and if these become shallow enough this could potentially lead to some resuspension of bottom sediments. The instream activities of animals can also be a major localised source of sediment resuspension, and affected waterholes can remain turbid for long periods if they are experiencing little or no flow. If there is strong baseflow present, plugs of turbid water may be carried significant distances downstream before the resuspended sediment either settles or disperses sufficiently to restore water clarity. However, many rivers support significant hyporheic flow and this can allow the bed sands to act as a filter, resulting in more rapid removal of suspended sediments.

Accordingly, if stormwaters have been fully displaced by groundwater driven baseflows, rivers run clear and they generally stay that way at least until flows fall to the point where they are no longer sufficient to wash away the contaminated water introduced by localised rain events or instream disturbances. Some rivers contain such large volumes of standing water that it can take a long time for baseflows to displace the turbid stormwaters detained within the system. In fact there are some rivers that never receive sufficient baseflow to accomplish this. Some systems of this kind may run clear only during wet years (when rainfall is high enough to recharge the groundwater aquifers that drive baseflows), and others may always be turbid. Many of the chronically turbid river reaches simply receive little or no baseflow, but there some reasonably high baseflow systems that behave this way because they contain very large waterholes. It must be stressed that these are reach scale effects; reaches located above large waterholes may run clear even though the waterholes and the reaches below them do not. Consequently it is rare to find turbid baseflows in upper stream reaches but quite common to encounter them in lower reaches.

Groundwaters generally leach significant amounts of mineral salts from soils and aquifer formation materials, so they are always more saline, and often harder (i.e. contain more calcium and magnesium salt) than surface runoff. The quality of this water depends on the hydrogeology of the groundwater formation and can vary quite widely even at local spatial scales. There are for example many small springs that occur in areas known to contain mineralogical anomalies of various kinds, including potentially economic reserves of heavy metals. Each of these could have distinctive and unusual water quality characteristics and would need to be assessed independently. However, most of the aquifers in question are small and low yield, so they are only likely to potentially influence a few small tributary streams. The baseflows in large rivers are generally driven by water from much larger regional aquifer systems with known geochemical signatures.

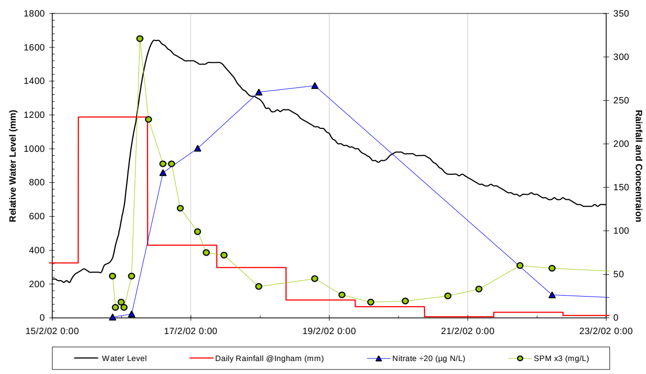
The highest and most reliable baseflows come from limestone aquifers. The hardness concentrations in these calcium and magnesium bicarbonate dominated waters may be high enough to promote colloid flocculation thereby enhancing water clarity and potentially removing trace contaminants such as metals and phosphate from the water column. Nutrient concentrations vary depending on the soil type of recharge areas, and the mechanics of recharge. As a general rule most groundwaters contain relatively high concentrations of nitrate (which is readily leached from soils because it does not generally sorb to the surface of soil particles) and low concentrations of phosphate (which tenaciously sorbs to the surface of many types of soil particles). The potential significance of this to receiving ecosystems is discussed in the next section.

Alluvial aquifers can also be a major source of strong baseflow, especially during the early dry season. Many such aquifers essentially contain sand-filtered river water, characterised by low concentrations of particulate contaminants (including nitrogen and phosphorus) and moderate ionic strength. The baseflows in some rivers are supplemented by other local and/or regional aquifer systems that are recharged by direct infiltration. The quality of these waters is difficult to predict because the catchments, formation materials and water residence times, all vary. Nevertheless, existing ambient water quality data for streams fed by such aquifers suggest that very few high-yield aquifers in this region contain water with EC concentrations in excess of 2000 µS $cm^{-1}$, and that most are considerably less saline than that.

A lot of the subsurface water that enters a stream on the early falling stages of event hydrographs can come from saturated riparian soils. This throughflow of soil-water occurs as soon as water levels in the stream begin to fall. This is not true baseflow and is seldom sustained for long.

However, in flat subcatchments with permeable soils the transition from throughflow to baseflow can be difficult to discern unless the soil water has distinctive water quality characteristics, as is the case in Figure 14 (Pearson *et a*l 2003a). In this figure daily rainfall is shown in red, stream height in black, SPM concentration in green and nitrate concentration in blue. This example has been selected because the monitoring site is located in an agricultural area where soils have been enriched with nitrate, giving the soil water throughflows a strong distinctive signature that clearly demarks the transitions from one water source to another.

**Figure 14: Variations in SPM and nitrate during an event in Lagoon Creek on the Herbert River floodplain (Qld).**



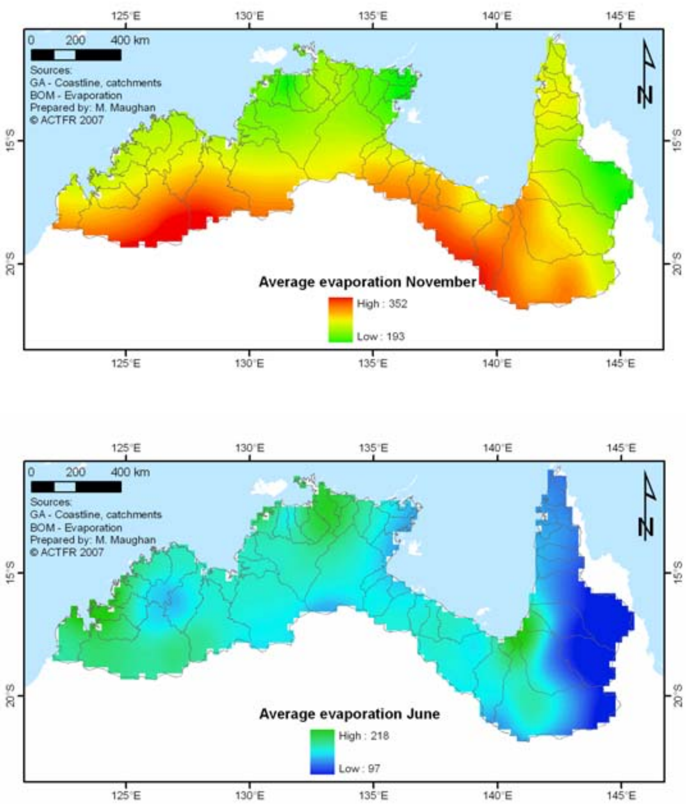

Since the site is influenced by agricultural runoff, the contaminant concentrations shown on the figure are not indicative of natural conditions. However, the hydrology of the site is still largely unaltered so the timing and magnitude of variations in the relative contributions from different water sources are still typical of natural systems, even though the quality of the water coming from each source are not. Ostensibly this figure provides a conceptual depiction of the water quality dynamics of many local streams; the green line representing turbid surface runoff and the blue line representing clear soil-water. The return to groundwater dominated baseflow is marked by the point at which the blue line falls to low levels. (Note that the gradual rise in turbidity that occurred during the transition to baseflow on this graph was accompanied by increased chlorophyll a concentrations and can be attributed to a minor phytoplankton bloom. This is also typical of what happens in many natural systems in the aftermath of a flow event).

## 2.4 Evapo-concentration

The water residence times of regional rivers always increase substantially as flow rates fall during the dry season, and especially when surface flows cease. Intermittent rivers with deep basal sands may sustain substantial hyporheic flows for months after gauging stations stop recording flows, so the water residence time of waterholes can easily be overestimated. Nevertheless most such waterholes do eventually stop flowing at some stage during a normal dry season.

The volumes of water lost due to evaporation increases with increasing water residence time, and in northern Australia evaporation rates are always very significant during the dry season; monthly values ranging from a minimum of 100 to 250 mm in June to a maximum of 200 to 350 mm in November, as can be seen in Figure 15. In practice some natural waters are exposed to the wind and evaporate more quickly than this, while others receive some shading and evaporate more slowly. There are also times and places where transpiration by riparian and aquatic vegetation can result in substantial water losses. Despite these variables, pan evaporation rates are still a useful guide for first-order prediction of evaporative water losses in most northern rivers.

**Figure 15: Mean monthly pan evaporation (mm) in June and November**




Since water residence times can extend to several months during the dry season, cumulative water losses can amount to a considerable volume. Most natural water quality contaminants are non-volatile and are left behind when water evaporates, so their concentrations increase as the water volumes decline.

This process, termed evapo-concentration, can significantly affect the quality of river waters, especially during the latter half of the year. The volume of water lost from a waterbody due to evaporation can be estimated by simply multiplying the millimetres of evaporation by the water surface area. However, in order to determine what effect this has on contaminant concentrations, it is necessary to know the volume of the waterbody, because concentrations increase by a factor that is inversely proportional to the percentage reduction in volume. This means that if evaporation rates remain constant, evapo-concentration effects increase substantially as the volume of a waterbody declines. For instance in a waterbody that starts out 3 metres deep, an evaporative loss of say 300 mm, could reduce the water level to 80% of its original volume meaning that contaminant concentrations rise by 25%. However, in a waterbody that is initially 600 mm deep the same evaporative water loss could reduce water volume to only 50% of its original value resulting in a 2-fold increase in contaminant concentrations.

Over the course of the dry season, water residence times and evaporation rates both gradually increase while water volumes decrease, accordingly evapo-concentration effects are distinctly non-linear and can increase dramatically towards the end of the year.

Most major dissolved ions are non-volatile and are not substantially affected by natural processes such as precipitation or assimilation. (Many of these parameters participate in biochemical reactions but in natural waters they are generally present in such excess and/or recycled so rapidly that overall concentrations are not significantly affected). Concentrations of these so-called conservative parameters, which include EC, chloride, hardness, and in some cases alkalinity, generally increase in accordance with predicted natural evapo-concentration rates. This process is often overlooked in water quality studies and as a result natural increases in EC levels have sometimes been misconstrued as being indicative of increased groundwater salinity and/or anthropogenic impacts.

Other water quality parameters such as nutrients and SPM are non-conservative, meaning that they are so strongly influenced by other biophysical processes that concentrations can still fluctuate independently. Nevertheless, at times and places where the concentrations of conservative parameters are being driven mainly by evapo-concentration (and during the latter half of the dry season that would include most waterbodies), trends in parameters such as EC can be used to predict what the concentrations of non-conservative parameters would have been if they had not been affected by other processes. This can be a very useful tool for assessing the significance of variations in concentrations of parameters such as nutrients and SPM.

In flowing systems water constantly evaporates as it flows downstream so it is not uncommon to find that EC is strongly correlated to the distance from the source. The flows in many of the river systems of eastern Australia are driven by mountainous headwater streams and/or groundwater springs located at the base of ranges. This means that much of the water in the lower reaches actually originates from points that are reasonably close to the river source. In such situations it is feasible to obtain reliably meaningful estimates of the distance from source, by simply consulting a topographic map. However, this is not a strategy that can be employed in northern Australian catchments because the rivers are not headwater driven – rainfall is considerably higher near the river mouth than it is near the source, and the groundwater formations that drive baseflows are not nearly so constrained to one particular regional physiographic divide nor are they necessarily located anywhere near the source. In this region meaningful estimates of distance from source will be contingent on specific knowledge of the main water sources in each individual river system.

## 2.5 Nutrient and Contaminant Partitioning

The limnological heterogeneity discussed in section 2.2 is indicative of spatial and temporal partitioning of biological and chemical water quality processes. In fact it is difficult to think of an ecologically important water quality process that is not at least indirectly dependent on pH, temperature and/or DO availability.

The development of a hypoxic sediment-water interface for instance can transform bottom sediments from a nutrient sink into a nutrient source. It can also either substantially decrease or increase denitrification rates depending on circumstances. Denitrifying bacteria are facultative anaerobes that can obtain their oxygen from nitrate when they need to (converting nitrogen into non-bioavailable gaseous forms in the process). They only do this when DO is in short supply, hence severely hypoxic bottom sediments can potentially provide an ideal substrate for denitrification. However, the process can only occur if nitrogen is available in the form of nitrate. Organic nitrogen can be broken down to form ammonia (termed ammonification) under all oxic conditions but subsequent conversion to nitrate (termed nitrification) is only possible if there is adequate DO available to oxidise the ammonia. The heterotrophic microbes that perform these nutrient conversions require a constant supply of organic carbon and generally attach themselves to solid surfaces. Accordingly the reaction rates sustained within the benthos are generally orders of magnitude higher than they are in the open water column.

Diel periodicity creates potentially ideal circumstances for nutrient cycling, with DO concentrations rising sufficiently to facilitate nitrification during the day and then falling to levels low enough to support denitrification during the night. However, the spatial partitioning of these temporal variations is crucial to outcomes. If the water column is stably stratified, diel periodicity may be restricted to the near-surface water layer where nutrient conversion rates are limited not only by the lack of available microbial substrates, but also uptake of dissolved nutrients by phytoplankton (which can rapidly assimilate nitrogen in the form of either ammonia or nitrate). In this case bottom waters never mix, so they are unable to obtain DO or nitrate from the surface layer. If these waters are aphotic they usually remain constantly hypoxic, and could potentially support high denitrification rates, but they are unable to do so because they cannot replenish depleted nitrate supplies.

Photic bottom waters and benthic habitats shallow enough to occupy the mixed surface layer have the potential to sustain high nitrification and denitrification rates. However, these waters also support very high levels of autotrophy, so nitrate and ammonia can be in short supply due to assimilation.

In lentic systems microbial nutrient cycling is likely to be most efficient in the bottom waters of diurnally stratified sites because overnight mixing allows the bottom waters to replenish nitrate supplies while daytime stratification allows them to become hypoxic enough to support high levels of denitrification. This can occur in lotic systems, however, when flows are present the same effect can also occur when water flows from one different type of waterbody to another, for example from a shallow run or riffle to a deep stratified pool.

The effects of microbial cycling on ambient nutrient concentrations are usually only evident in aphotic waters. In photic waters plants and/or phytoplankton generally assimilate bioavailable nutrients so rapidly that it can be difficult to resolve the effects of other processes.

For example the ACTFR (contract research reports[1]) studied some key aspects of the nutrient dynamics of tributary streams in the Gregory River (Queensland). They found that, even though flows were fed from limestone aquifers that contained around 750 µg N $L^{-1}$ of nitrate, concentrations in the river system under baseflow conditions were consistently less than 5 µg N $L^{-1}$. More detailed water quality monitoring carried out in the vicinity of identified inflow points confirmed that the nitrate concentrations in the streams rose substantially as the water passed inflow points, but assimilation rates were so high that virtually all of the nitrate had been converted to organic nitrogen by the time that the water had traveled a short distance downstream. Actual travel times and distances varied over time and between sites but as a general guide, the nitrate levels in a tributary that was carrying baseflow to the river 80 km further downstream were typically reduced to undetectable levels within 0.8 to 1.8 km of the springs that were driving most of the flow.

This rapid nitrate assimilation was consistently observed whenever streams were at or near baseflow, however, the dynamics of total nitrogen concentrations varied considerably depending on circumstances. During the early dry season some sites remained sufficiently turbid to inhibit benthic productivity. In these instances nitrate was assimilated by phytoplankton which were carried downstream meaning that the total concentrations of nitrogen delivered to the rest of the river system remained elevated even though the concentrations of dissolved inorganic nitrogen species were reduced to negligible levels. (The fact that total nitrogen concentrations remained elevated appeared to suggest that denitrification was not a significant nitrate removal mechanism. However, it should be noted that sampling was only carried out during daylight hours when DO levels were high enough to inhibit denitrification. As is usually the case in this region DO levels fell dramatically overnight and it is possible that significant denitrification would have occurred at such times.)

The situation changed considerably once turbidity fell to low enough levels for benthic autotrophy to become established, and in aquatic macrophyte dominated sites such as Lawn Hill Gorge, this can happen very soon after wet season flows subside. In this case nitrate was incorporated into the biomass of attached photosynthetic organisms resulting in the removal of nitrogen from the water column. Consequently the total nitrogen concentrations in the water being carried downstream were substantially reduced, often to barely detectable levels. (Again it is highly likely that denitrification also plays a significant role at night but this has not been confirmed.)

The rapid nitrogen assimilation rates were a little surprising because the groundwater contained very low concentrations of phosphorus, as did the receiving streams when they were running clear. Turbid streams contained similarly low concentrations of dissolved inorganic phosphorus, and although particulate phosphorus levels were sometimes significant, there still did not appear to be sufficient available phosphorus to fuel such rapid uptake of nitrogen.

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[1] Access to these reports is subject to approval by commercial clients. Contact ACTFR for details.

However, closer examination showed that the pore waters in bottom sands contained more than enough available phosphorus to explain the observed productivity rates. It was therefore hypothesized, though not proven, that the streams were shallow and turbulent enough for phytoplankton to be able to exploit this benthic nutrient pool. This seems a reasonable hypothesis given that the streams in question are quite shallow along most of their length, and contain permeable basal sands capable of supporting significant hyporheic flow (which would promote the infusion of pore waters into the overlying water column). Regardless of the precise reasons it is clear that the concentrations of nutrients in the water column were a very poor predictor of nutrient availability.

In fact in our experience it is impossible to meaningfully interpret the significance of ambient nutrient concentration data unless the hydrology, limnology and ecology of sites are sufficiently well understood to be able to infer how nutrients are spatially and temporally partitioned. This could be facilitated by applying a relatively simple typology to differentiate between waters with contrasting limnological and ecological traits.

As a starting point we propose that waterbodies could be classified into 3 limnological categories: mixed, partially mixed and stratified. Note that this classification applies to a waterbody as whole and that the classification of most waterbodies is very likely to change over the course of a typical year, with many instream waterholes for example, being mixed during periods of high flow, partially mixed during the falling limb of the hydrograph and stratified during the late dry season. Note that in this context the term “mixed” relates only to the water and associated solutes – coarse particulate materials, especially those with a diameter greater than 63 microns, are rarely homogenously mixed throughout the water column, although the very fine particulate contaminants (such as clays and colloids) which are of principle interest in most ecological water quality investigations often behave more like solute. Waters should not have to be completely homogenous to be classified as mixed, but they should be sufficiently turbulent to ensure that vertical temperature gradients do not develop. Waters in this category may not be laterally mixed. For example even during floods when conditions are extremely turbulent, water discharged from a tributary stream into a wide river reach can travel very large distances downstream before mixing.

Partially mixed waters would include sites that exhibit diurnal stratification as well as those that stratify and destratify at more irregular time intervals. For example some waters mix when the weather is windy but stratify when conditions are calm. Note that it is only necessary to carry out depth profiles during daylight hours on a day when conditions are very calm in order to ascertain if a waterbody stratifies or not. However, to find out if it is diurnally stratified, it is necessary to carry out additional measurements during the very early hours of the morning.

Broadly speaking the water contained within any of the above waterbody types can be further classified into 3 distinct functional categories: 1) heterotrophic; 2) photic-limnetic and 3) photic-benthic.

Heterotrophic waters support relatively low levels of primary production due mainly to inadequate light availability in the water column. Many streams contain water of this kind during and soon after the wet season when turbidity levels are still quite high, standing water levels are elevated and instream autotrophic biomass has not had time to recover from the wet season flush-out. During the dry season heterotrophic waters are most commonly encountered in the aphotic bottom layers of stably stratified waterbodies, although the dry inland catchments contain some waterbodies that remain largely heterotrophic for most of the year. Since the productivity of these waters is not limited by nutrient availability, changes in nutrient concentration are often largely inconsequential, provided of course that ammonia concentrations do not increase to the point where they become toxic to aquatic fauna (in this region no other nutrient occurs at high enough concentrations to become toxic).

Significant ammonia concentrations can periodically develop in heterotrophic bottom waters because they are generally too hypoxic to support nitrification. Ammonia is about 50 to 100 times less toxic at low pH than it is at high pH, and since heterotrophic waters are normally moderately acidic or circum-neutral, toxicological problems are not a frequent occurrence. However, as discussed previously, the pH levels in photic surface waters can be very high, so there is potential for acute problems such as fish kills to occur if and when the waterbody destratifies.

The heterotrophic waters that occur in flowing streams at the end of the wet season are often reasonably well-mixed therefore traditional near-surface grab sampling techniques can usually yield reasonably representative contaminant concentration estimates. Near-surface waters usually only remain heterotrophic during the dry season if they are very turbid, and since turbid waters are highly prone to vertical stratification, it is not feasible to collect discrete grab samples that are representative of more than a small portion of the water column. Regardless of the clarity of the overlying waters, heterotrophic bottom waters generally contain elevated concentrations of dissolved inorganic nutrients and bioavailable trace metals. Historically these waters have not been sampled in routine ambient water quality monitoring programs so they are not represented at all in existing databases.

Under the proposed nomenclature, “photic-limnetic waters” are those that admit sufficient sunlight to support significant levels of primary production within the water column but not within the benthos. As mentioned above there are some waterbodies in the dry inland catchments that always remain sufficiently turbid to prevent the water column from becoming autotrophic, but at some stage of the annual hydrograph, most regional streams become sufficiently clear to support significant levels of phytoplankton productivity. In waterbodies that retain benthic biomass in the aftermath of wet season flushes (as is often the case in systems that host assemblages of deep-rooted aquatic macrophytes) this may be a relatively brief transitional state that occurs during the early falling limb of the hydrograph, while water depths and turbidity levels have not yet fallen sufficiently for benthic productivity to be re-activated. However, many streams are so well flushed by wet season flow events that it can take months for benthic biomass to build up to significant levels even after photic conditions have been restored. Even so, at some stage during the dry season most shallow stream reaches begin to support significant benthic productivity and at such times photic- limnetic waters are mostly confined to the mixed surface layers of deep stratified waterbodies.

The nutrients contained in limnetic waters are assimilated mainly by microscopic phytoplankton that spend most of their time suspended in the water column. Accordingly there is some chance that they will be representatively captured in water samples, meaning that total nutrient concentrations can be indicative of the nutrient pool associated with the standing crop. In practice many local waters are so poorly mixed that phytoplankton communities are often dominated by cyanobacteria and these migrate through the water column by constantly adjusting their buoyancy. Hence the prospects of actually obtaining representative grab samples from lentic waters are not always good.

Photic-benthic waters are those that regularly come in direct contact with photosynthetically productive benthic habitats. The waters in many moderately shallow stream reaches fall into this category most of the time, and by the end of the dry season, even quite deep waterholes usually contain extensive zones that support significant levels of benthic productivity. The shallow littoral zones of most stratified waterholes contain photic-benthic waters and in many cases even the poorly mixed bottom waters receive enough light to fall into this category. Biomass accumulation and turnover rates are usually considerably higher in the benthos than they are in the water column, so the effects on water quality can be substantial. Benthic plants and algae remove nutrients and other bioavailable contaminants from the water column; hence, as was the case in the Gregory River example discussed earlier, the total nutrient concentrations obtained in water sampling programs can actually be inversely proportional to benthic productivity rates – i.e. the most productive waters report the lowest nutrient readings.

Benthic autotrophs are also able to gain access to a sedimentary nutrient pool that is not freely available to limnetic organisms, and consequently the productivity rates in these waters can seldom be predicted by measuring surface inputs alone. In such situations it is traditional to recommend alternative monitoring strategies such as determining cumulative nutrient loads and/or analysing the nutrient content of bottom sediments. However, as will be discussed later, this can be an enormously complicated and logistically difficult research exercise, and is certainly not a viable tactic for routine monitoring and assessment applications.

There is intense competition between limnetic and benthic autotrophs in photic-benthic waters. In some streams baseflows are strong enough to constantly wash away phytoplankton so they are rarely able to maintain sufficient biomass to successfully compete with attached benthic species. In such cases benthic biomass can often build up to the point where nutrients are removed from the water column so efficiently that phytoplankton productivity is severely nutrient-limited. If this happens phytoplankton biomass may no longer be able to accumulate in the water column even if flow ceases. Nevertheless, the water residence time in some large waterholes and deep river reaches may be sufficient to allow significant phytoplankton biomass to accumulate even when baseflows are strong, and there are also many waterholes that stop flowing before benthic productivity has had time to re-establish.

These waters often maintain quite significant concentrations of phytoplankton (and therefore total nutrients) for most of the dry season, and even though the concentrations of dissolved inorganic nutrient in the water column are generally extremely low, quite high turnover rates are maintained (as evidenced by the strong diel periodicity of DO and pH). In lentic waters allochthonous nutrient inputs are negligible for most of the dry season, so this rapid recycling does not normally lead to significant increases in biomass and in many cases phytoplankton concentrations gradually decline. In contrast, due to the ample nutrient reserves contained in bottom sediments, benthic biomass continues to increase for as long as light levels at the benthos remain adequate. However, later in the dry season localised instream disturbances and/or inputs of runoff from pre-flush events can cause sudden and often dramatic reductions in light availability, and this not only induces DO sags, as discussed in section 2.2, but also substantially alters nutrient processing and partitioning.

Pre-flush runoff carries nutrients into the water column but can be sufficiently turbid to initially inhibit photosynthesis. Accordingly the water column becomes heterotrophic for at least a while, and in dry inland subcatchments that contain a lot of erodible clay soils, streams may sometimes remain that way until a flushing event occurs – and during droughts that can be a considerable time. Due to decomposition of allochthonous organic matter, pre-existing plant biomass and sometimes, fauna killed by DO sags, concentrations of dissolved organic nutrients can rise to quite high levels. At the same time silt particles gradually begin to settle and as a result turbidity often falls sufficiently for the upper layers of the water column to become photic again. This provides ideal conditions for rapid growth of phytoplankton, especially cyanobacteria, and for this reason quite severe blooms have often been observed in waterholes towards the end of the dry season.

Our field observations and laboratory experiments suggest that common local submergent plant species such as Ceratophyllum gradually decompose when kept in the dark but they can still remain viable for about a month, exhibiting very rapid re-growth as soon as light comes available again. Photic conditions may sometimes be restored to the benthos within this time frame, particularly in shallow waters that have temporarily become turbid due to resuspension of bottom silt. However, the runoff introduced by pre-flush events usually contains sufficient colloidal material to prevent complete clarification from occurring, and it also often significantly increases the depth of the water column, making it more difficult for light to reach the bottom. It is therefore rare for benthic productivity to be fully restored before wet season rains arrive.

Waterbodies that support significant stands of various types of aquatic macrophytes may alter the processes discussed in the preceding paragraphs. For example stands of tall submergent and/or semi-emergent macrophytes can form canopies that extend well into the water column, sometimes reaching the surface. Even in reasonably deep waters these, and the epiphytic periphyton growing on them, can actually take over from phytoplankton as the major limnetic producers. For example in the Ross River in Townsville, extremely dense stands of Hydrilla and Ceratophyllum regularly manage to dominate the surface water layer of weirpools that are 4 to 8 m deep, and total nutrient concentrations in the water column can be reduced to negligible levels under such circumstances. These plants are very well positioned to recover from brief episodes of turbidity, so phytoplankton blooms do not commonly occur at such sites.

Emergent and floating plants can be dominant features of many off-channel wetlands but since they are not major floral components of many natural riverine wetlands, they will only be cursorily mentioned here.

Emergent and floating plants respire and photosynthesise by exchanging gases with the atmosphere - they do not directly influence the DO or carbon dioxide concentrations in the water column in the same way as submerged plants. However, their indirect affects on dissolved gases, productivity and nutrient partitioning within the water column can be substantial. Dense assemblages providing shade, inhibit re-aeration and mixing, and deliver constant inputs of detrital organic matter, substantially increasing the heterotrophy of the underlying water column, and leading to the development of hypoxia and elevated concentrations of dissolved inorganic nutrients. The effects of smaller and/or less dense assemblages are much more subtle. Sparse stands of emergents such as reeds may admit sufficient light for photosynthesis to occur, and can provide substrate for periphyton allowing high levels of autotrophy to be maintained within the water column. Most native floating macrophyte species are relatively small and do not strongly influence the water column unless present in very dense assemblages. However, unlike other emergents, they must obtain nutrients from the water column and therefore compete with phytoplankton and emergents.

Accordingly, the water underlying very dense stands of emergent and/or floating macrophytes often contain high concentrations of nutrients, mainly in dissolved forms, while the water under less dense stands may be contain unusually low concentrations of both dissolved and particulate nutrients.

Traditional water quality monitoring and interpretation methods are generally premised on the assumptions that high concentrations of nutrient in the water column are indicative of eutrophic conditions, low concentrations are indicative of oligotrophy, and total nutrient concentrations provide some indication of existing nutrient availability. Of all the possible combinations of water and waterbody types proposed in this section, only one combination goes close to complying with these assumptions, that being photic-limnetic waters that occur in mixed waterbodies. However, in this region there are many riverine waterbodies that only mix for a very short period during high flow events and most of them are heterotrophic for much of that time. Very few riverine waterbodies remain mixed during the dry season and most of those that do are shallow and would be classified as photic-benthic. Basically, in this region the concentrations of nutrients and most other bioactive contaminants contained within the water column of freshwater habitats are so strongly influenced by localised and highly variable limnological factors that they cannot be directly linked to productivity.

Most texts including the ANZECC and ARMCANZ (2000) national water quality guidelines recommend dealing with these situations by assessing nutrient loads rather concentrations. However, given the immense temporal variability of regional aquatic systems it would be a major undertaking to do this at just one study site and there are many sites where the prospects of a successful outcome are currently negligible. For a start many sites are so inaccessible that it can be logistically difficult and expensive, not to mention hazardous, to conduct a single field visit, let alone mount a major long term investigation. For example in the study referred to earlier in this section, sites could only be accessed by helicopter meaning that work could only be carried out when weather conditions were safe and at sites in the vicinity of safe landing areas. Moreover many of the field techniques that are routinely employed in other parts of the world cannot be used safely in the crocodile infested waters of northern Australia. Logistics aside there are also some significant scientific challenges to overcome.

In temperate rainfall regions it is often feasible to use annual and/or seasonal nutrient export loads (i.e. cumulative nutrient fluxes) as an indicator of internal nutrient loading (i.e. the amount of nutrient detained and/or utilised by instream ecosystems), however, due to the flashy nature of hydrographs in this region there is an enormous disparity between export loads and internal nutrient loads. As can be seen in previous figures the vast majority of water and associated contaminants pass through most systems in just a few days (in small streams) to a few weeks (in large streams) during major wet season spates.

The quantities of water, and therefore nutrients that pass through the system during the rest of the year (i.e. most of the time) are often negligible compared to these brief storm pulses (Butler 2005, Butler and Burrows 2006)). There may be some retention of stormwater in off-channel waterbodies and/or deposition of particulate contaminants on river floodplains, but the coarse-grained sedimentary materials left behind in the streambeds of most streams in the aftermath of wet season flow events bear testament to the fact that there is very little retention of water quality-related contaminants within the main river channels.

In fact most of the instream biomass that accumulated over the annual drought is also washed away during these events, leaving many benthic habitats in a relatively depauperate condition - ostensibly the ecosystem is reset. The falling limb of the hydrograph associated with the last major wet season event each year initiates an annual recovery process that is often characterised by a series of ecological successions involving major fundamental changes in the ways that the ecosystem responds to nutrient inputs. On the falling limb of the hydrograph many systems contain turbid heterotrophic water, so most nutrients may pass through without being assimilated. There may be some minor deposition of particulate nutrients in quite backwaters, but most of the SPM that remains in suspension at this stage of the hydrograph is too fine to settle before the water passes through. As noted earlier there are some systems (especially in dry inland catchments) that remain in this state most of the time, but in other systems (especially in wet near-coastal catchments) it can be such a transitory event that it is difficult to detect. On the tail of the hydrograph most large perennial systems are photic-limnetic meaning that nutrients are being assimilated within the water column but are still being carried downstream in the form of plankton and organic detritus. By the time stable baseflow has been restored many parts of the benthos have become photic and attached autotrophs begin assimilating and detaining nutrients. Since there is usually very little, if any, pre-existing autotrophic biomass remaining in the benthos, assimilation rates can initially be very slow and for quite some time may be difficult to detect.

There is very little if any visible surface flow in most regional streams by the time benthic biomass has increased to the point where the nutrient concentrations in the overlying water column are being noticeably affected. However, many systems support significant hyporheic flow, and it is highly likely that this cryptic, and rarely monitored subsurface input source plays a major role in the nutrition of instream ecosystems and especially benthic autotroph communities. Once flow has stopped, as it does in most streams, the waterbodies are effectively decoupled from most of the catchment and subsequent additions to the available nutrient pool are dependent on instream processes (such as nitrogen fixation and releases of sediment bound nutrients) and inputs from random localised events. The latter include not only pre-flush rain events but also direct inputs of excrement from terrestrial animals that periodically congregate around waterholes. In most cases the loads of nutrients introduced into waterbodies during the latter part of the wet season are extremely small, even compared to the relatively modest amounts that pass through the system under baseflow conditions. Nevertheless, as discussed previously, they can have a remarkable impact on the ecology of these small lentic systems, sometimes even leading to the development of conditions that are acutely lethal to aquatic fauna. Ostensibly the ultimate fate of most of the biomass produced during the entire dry season may be contingent on just one or two relatively minor random events, the outcomes of which depend at least as much on instream limnological factors as they do on nutrient loading.

Basically, in these systems, it is extremely difficult and often logistically impossible to obtain ecologically meaningful estimates of nutrient loading, and even more difficult to see how this information could be used to accurately predict ecological outcomes.

# 3. SALINITY AND IONIC COMPOSITION

Workshop discussions initially focused on salinity and related ionic constituents such as alkalinity and hardness – mainly because they are by far the most well-represented parameters contained within existing regional databases (see appendix). This study focuses primarily on the freshwater parts of river systems, but in practice it is not always obvious if sites located near the coast are sufficiently free of tidal influences to be classed as freshwater habitats. Moreover, some waterbodies provide freshwater habitats during the wet season but are dominated by marine influences later in the dry season. In northern Australia fresh waters can be more than a thousand times less saline than seawater, so even quite minor tidal incursions can have an enormous effect on both water quality and aquatic ecosystems. Accordingly, a geospatial classification scheme capable of indicating when and where marine influences are likely to become significant would be very useful, especially in the macrotidal systems in the northwest where the tidal wedge has the potential to penetrate long distances upstream.

*Under international conventions (APHA, 2005) salinity is a unitless measure of the ionic strength of seawater. Estimates are often calculated from electrical conductivity values using standard formulae derived from empirical analysis of international seawater standards. Most modern conductivity meters perform these calculations automatically. Conductivity-based salinity estimates are measured on an arbitrary scale, termed "the practical salinity scale", consequently results are often reported in practical salinity units (psu). Oceanic seawater has a salinity of approximately 35 psu. Although the term salinity is used colloquially in reference to the salt content of all waters, salinity measurements are only valid for waters with ionic ratios equivalent to that of seawater, so they should not be used in fresh waters. In order to avoid confusion some authors employ the term halinity when specifically referring to the salinity of marine waters.*

*A wide variety of alternative analytical parameters and methods are used to estimate the salt content of fresh waters, the most popular being simple direct measurement of electrical conductivity (EC) to yield results in* $\mu S\ cm^{-1}$. *Empirical formulae are often used to convert EC values into total dissolved salt (TDS) concentrations (*$mg\ l^{-1}$*) but because ion ratios vary considerably over time and space in fresh water, this method is not totally reliable.*

## 3.1 Tidally Influenced Waters

The salinity levels in near-coastal river reaches, and the wetland systems associated with them, range from virtually zero to greater than 60 psu. In fact in places where tidal waters intercept freshwater flows, a large proportion of this variability range can be observed over the course of a single tide cycle, and the full possible range of values can be encountered over the course of a normal year. Since seawater is much denser than freshwater, mixing is very inefficient, and as a result there are occasions when freshwater and seawater can coexist in different parts of the same waterbody. Moreover, due to the seasonality of rainfall and stream flows in this region, many water bodies that function as freshwater habitats at one time of the year can become saline, or even super-saline, at another. The time scales involved and the magnitude of the effects vary enormously between sites. At one extreme there are many estuarine river reaches that may only experience freshwater conditions for a few weeks each year during the peak of large flood events. At the other extreme there may be reaches where the upper water column is always fresh but the benthos is periodically exposed to seawater for brief periods during the largest spring tides. Transient salinity fluctuations of these kinds can have persistent and far-reaching ecological consequences, because there are many specialised freshwater organisms that are unable to survive even brief exposures to seawater, and some marine organisms that are unable to tolerate low ionic strength water.

Salinity regime is therefore a critical determinant of habitat utilisation and diversity in near-coastal wetland systems, and is the main factor responsible for ensuring that different kinds of estuarine habitats retain a distinctive ecological character. However, the spatio-temporal variability of such systems can be so substantial that the task of accurately determining the salinity regime of even a single waterbody can be a major undertaking. Accordingly it is unlikely to ever be logistically feasible to include detailed assessments in routine and/or broad-scale monitoring programs. Indicative salinity measurements obtained from less

intensive monitoring programs can still be of some use, but they are unlikely to provide an adequate basis for determining the salinity regimes and are usually only available for a very limited number of sites.
However, some of the key biophysical characteristics of a wetland, such as its vegetation, are sufficiently sensitive to salinity and/or tidal movement to potentially serve as indirect indicators of salinity regime. In order for these indicators to be successfully employed in broad-scale wetland classification applications, they need to be amenable to assessment by remote sensing techniques. Potential indicators include temporal variations in riparian and aquatic vegetation assemblages, soil/sediment condition, the total area of salt flats contained within tidal catchment areas, and potentially at least, water colour and/or turbidity. (The latter being relevant because colloidal sediments, which remain dispersed in fresh water, flocculate upon contact with seawater). To date indicators such as these are mainly used as qualitative aids to guide professional judgement, so existing classification procedures are highly subjective and can only be carried out by experienced specialists. Further research into the precise relationships between salinity, indicator variables and remote sensing data is needed in order to develop more quantitative methods.

The expert panel concluded that, although virtually all water quality indicators play some supporting role, the ecology of most freshwater habitats in northern Australia is primarily governed by two main water quality-related variables – water transparency and dissolved oxygen (DO) availability. Salinity regime (often measured as electrical conductivity or EC) is a fundamental determinant of whether near-coastal waters should be considered to be, and is a valuable indicator for differentiating estuarine habitat classes.

## 3.2 Fresh Waters

The salinity levels in inland river reaches (i.e. those that are located upstream of tidal/marine influences) are always so much lower than seawater that they must be assessed on entirely different scales – for example, fresh water with a salinity twenty times lower than seawater would be considered to be highly saline and would present a potential threat to sensitive species.

Available evidence (see appendix) indicates that the vast majority of inland waters in this region are characterised by very low to moderate salinity levels, and concomitantly, low to moderate concentrations of all major ionic constituents such as alkalinity, hardness and chloride. Only a handful of the hundreds of freshwater sites represented in existing water quality databases report any EC values greater than 2000 µS $cm^{-1}$, and most inland catchments contain no sites with maximum values higher than about 800 µS $cm^{-1}$. Available toxicological data indicate that elevated salinity levels are unlikely to substantially impact on local freshwater organisms until EC levels significantly exceed 1500 to 2200 µS $cm^{-1}$ (see for example Dunlop *et al* 2008, Kefford *et al* 2007a, 2007b, 2006, Horrigan *et al* 2007, Marshall and Bailey 2004, Nielsen *et al* 2003a, 2003b, Hickey *et al* 2008, Mount *et al* 1997, Zalizniak *et al* 2006). Nevertheless, there is evidence that very low EC levels (<50 to 100 µS $cm^{-1}$) and/or very low concentrations of related ionic constituents such as calcium, can significantly alter macroinvertebrate community structures, presumably through non-toxicological effects such as nutritional deficiencies (see for example Horrigan *et al* 2005). Moreover, moderate concentrations of water hardness and alkalinity are beneficial to most ecosystems, as they reduce the bioavailability of several commonly occurring toxicants and in the case of alkalinity, help to stabilise pH values. Very soft low ionic strength waters are therefore potentially more vulnerable to a variety of water problems.

Effects of this kind are important and should not be ignored, but for classification purposes they are secondary to other more fundamental functional traits. For example, at the concentration levels in question here, EC may play a role in determining which species occupy a niche within a freshwater ecosystem, but it would rarely be expected to actually create or destroy niches, or to substantially alter the dynamics of energy and carbon flow. EC and other related parameters would be more appropriate to use at the sub-class level of a function-based classification scheme. However, this could only be done if there were sufficient water quality data to be able to properly assess spatio-temporal variability, and as discussed later in this report, existing data are not nearly extensive enough to meet this requirement, and there is little likelihood of this situation changing substantially in the foreseeable future.

# 4. WATER TRANSPARENCY

Water transparency is sometimes assessed directly by using meters to measure PAR attenuation through the water column, or indirectly by measuring vertical and/or horizontal sighting distances with black, or black and white (Secchi) discs. However, measurements of these kinds have traditionally been carried out almost exclusively in conjunction with specialised limnological research projects with limited regional scope. Since they have rarely if ever been included in broader regional water quality monitoring programs, knowledge of the water transparency variability patterns in most systems is semi-quantitative at best. There are some data available for parameters that are very closely related to water transparency – namely SPM, turbidity and in a few cases, colour. However, existing data are spatially and temporally patchy, of questionable or unknown reliability, and in the case of colour, extremely sparse. Colour is normally only a significant factor in river reaches that are influenced by large off-channel wetland assemblages and/or unusually dense assemblages of instream aquatic or semi-aquatic macrophytes, so in most inland systems, water transparency is mainly governed by SPM.

*Ecologists should note that the terminology used here is consistent with current conventions in water quality science rather than biology. SPM is notionally equivalent to seston. In water quality programs the phytoplankton portion of seston is sometimes estimated by conducting chlorophyll analyses, and on rare occasions the inorganic and organic portions of the seston are determined by conducting a slightly modified version of an ash free dry weight analysis to yield estimates of inorganic and organic SPM. However, none of the standard water quality analyses in current use discriminate between plankton and tripton, or the living and detrital components of the tripton.*

*In water quality science turbidity has a far more specific meaning than the colloquial definition; it relates to a family of standarised methods that use instruments called nephelometers to measure the amount of optical back scatter that occurs when a sample or a small part of the water column is illuminated by an artificial light source. Results are reported in arbitrary nephelometric turbidity units (NTU).Turbidity is a property that cannot be directly related to known ecological or limnological processes, nevertheless it can sometimes be an effective surrogate measure of SPM.*

*Colour is a property that is most commonly determined empirically, by comparing samples with internationally-accepted arbitrary standard solutions that generate a yellow-brown colouration of known intensity. Results are reported in Hazen colour units (HCU) or simply colour units. The analysis can be conducted on un-clarified samples to yield an "apparent colour" estimate or clarified samples to yield a "true colour" value. In this paper the term colour applies to true colour – i.e. that caused only by dissolved substances.*

*Water transparency is a generic term encompassing a range of parameters that can be used to estimate optical attenuation. These include but are by no means limited to Secchi depth, euphotic depth and light attenuation co-efficient.*

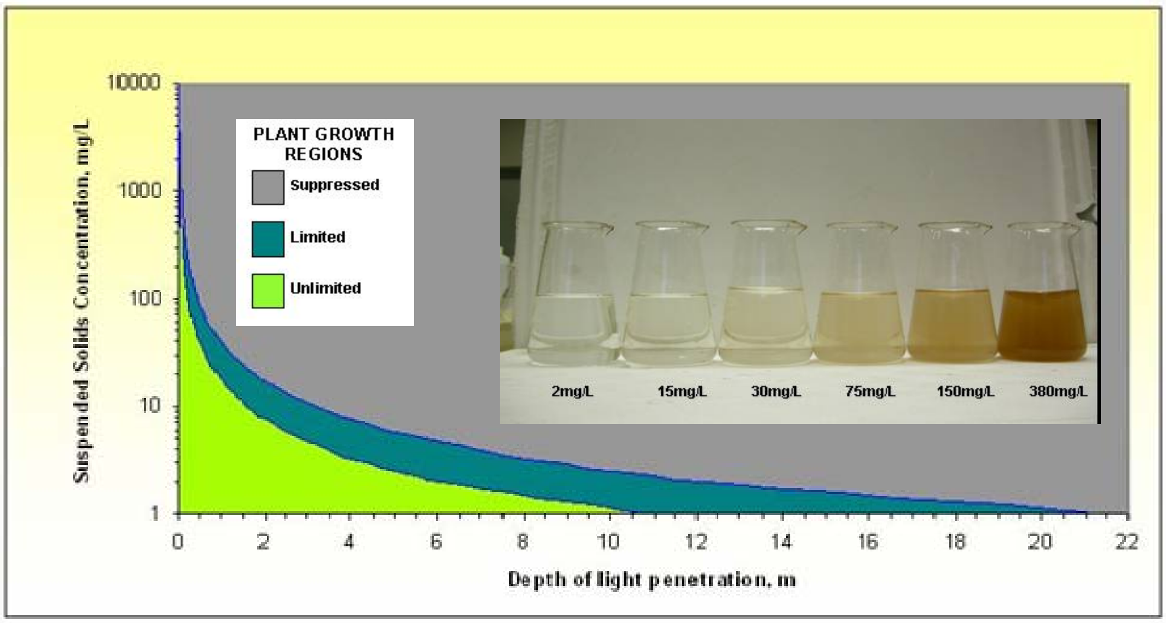
Turbidity and water transparency depend not only on the amount of SPM in the water but also the size, shape, light absorption and light scattering properties of the suspended particles. The SPM in storm water comprises mainly fine inorganic sediment particles, and although the size and consistency of the particles varies somewhat with flow, there is normally a reasonably reliable linear correlation between SPM and turbidity during storm events. Accordingly turbidity has often been successfully used as a surrogate for SPM in semi-quantitative studies of the dynamics of fluvial sediment transportation processes. However, as discussed previously, during the falling limb of the hydrograph water begins to enter rivers from many different sources containing different types of SPM. By the tail of the hydrograph SPM concentrations have often fallen by orders of magnitude while instream biological productivity has generally increased substantially.

Autochthonous particulate organic matter such as plankton can comprise an increasingly significant and sometimes dominant proportion of these smaller amounts of particulate material that remain suspended in the river at that time. Since these particles have quite distinctive and varied optical properties the correlation between SPM and turbidity can become so complex that it is extremely rare to find a reliable working relationship between the two parameters once baseflow has been restored. This is especially true in cases where SPM concentrations are below 10 to 40 mg/L. For example the $R^2$ values obtained from linear regression analysis of the turbidity and SPM data contained in the Queensland northern rivers database were greater than 0.87 if all data were used, but ranged between -2.5 and 2.7 when only baseflow samples with turbidities of less than 10 to 40 NTU were used (Butler 2005).

The correlations between these parameters and transparency are even less reliable, because the relationships between SPM/turbidity and transparency are distinctly non-linear, as shown in Figure 16 (Pearson *et al* 2003). This figure was produced by carrying out serial dilutions on one particular sample of natural SPM – other samples containing different types of SPM yield different curves with the same general shape.

It can be seen that when SPM levels are very high, concentration variations as large as hundreds of mg/l do not affect water transparencies enough to significantly alter ecological productivity – for example it would rarely matter much if euphotic depth changed from 0.005m or 0.01m because basically, transparency is still very low compared to the river's depth. Conversely when SPM levels are very low, a very small variation can alter the depth of light penetration by an amount that is very significant compared to the depth of a river – for example a variation of considerably less than 1 mg/L could alter the euphotic depth by more than a metre, which is very significant for the many river reaches that are less than a few metres deep.

**Figure 16: The relationship between suspended solids concentrations and water transparency**




It might nevertheless be possible to use SPM and/or turbidity data to classify waters (not waterbodies or sites) into one of at least three broad qualitative transparency-related categories:

1. Opaque – i.e. water that is so turbid that most of the water column will be aphotic regardless of depth. In this case water colour will be irrelevant and sensitivity to changes in turbidity will be low. Waters of this kind will almost always fall into the heterotrophic class proposed in the preceding Section.

2. Marginal – i.e. water that is sufficiently transparent to support significant autotrophy within the mixed surface layer of deep waters and in shallow benthic habitats, provided that it is not significantly coloured. Sensitivity to turbidity changes will be high in shallow sites and moderate in deep ones.

3. Transparent – i.e. water that is so transparent that autotrophy is likely to occur in all but the deepest of benthic habitats, unless colour levels are quite significant. Sensitivity to turbidity fluctuations may be exceptionally high, potentially resulting in very large changes in the productivity and DO

status of the waterbody, and especially its benthos. Waters of this kind would almost always be of the photic-benthic type proposed in the previous section.

These classifications are very coarse but they at least provide some capacity to differentiate between waters that are at the extreme ends of the natural stream metabolism spectrum. More accurate results would obviously be obtained by basing classifications on direct measurements of light attenuation and/or by including consideration of water colour as well as turbidity. However, there are little or no data available for these parameters, and no prospects of sufficient data coming available in the foreseeable future. There are undoubtedly some sites, especially in the near-coastal zone, where this would often lead to misclassification. Nevertheless the main shortcoming with the suggested approach is that a very large percentage of waters will inevitably fall into the somewhat non-descript “marginal” category.

The main problem is that, except in cases where the water is either exceptionally clear or almost completely opaque, it is impossible to assess the ecological significance of ambient water transparency values without taking timing and bathymetry into consideration. Unfortunately there are virtually no bathymetry data available for regional river systems even though water depth plays just as important a role as transparency in determining how much of the water column and/or the benthos lies within the photic zone.

# 5. OPTICAL DEPTH

## 5.1 Optical Depth as an Indicator of Ecological Function

The interactions between transparency and depth can be integrated into a single parameter, termed optical depth. This can be done in several different ways depending on the measurement techniques employed and the type of site being assessed, but notionally at least, it can be defined as the ratio of the total water depth to the euphotic depth (i.e. the depth to which light can penetrate). Low ratios indicate that waters are shallower than the euphotic depth meaning that most of the bottom receives enough light to ensure that benthic productivity is seldom light limited. Waters of this kind are classified as optically shallow and would normally be of the photic-benthic functional type (proposed in section 4), provided that these optical conditions have persisted long enough for autotrophic biomass to become established within the benthos. Note that waters of this kind can actually be quite deep if they are highly transparent.

Waters with high depth to euphotic depth ratios, indicating that only a small proportion of the benthos ever receives sufficient PAR to support significant autotrophy, are considered to be optically deep. Waters of this kind can actually be quite shallow if they are extremely turbid and/or highly coloured.

Conceptually the transition from optically deep to optically shallow is marked by a relatively sharp and well-defined threshold – i.e. the point at which benthic habitats begin to receive sufficient light to support autotrophy. In practice, however, water depth varies widely throughout natural waterbodies and as a consequence so does optical depth. In fact, except in cases where the water is so turbid that transparency is negligible, most waterbodies will exhibit at least some significant spatial variations in optical depth. These variations need to be analysed and parameterised in order to assess the optical characteristics of the waterbody as whole.

In order to do this it will be necessary to know both the euphotic depth and the bathymetry of each waterbody. Unfortunately existing regional databases contain almost no information of this kind, so the methods proposed here could not be implemented immediately. They could, however, be employed in the foreseeable future if the recommended monitoring is carried out, and we believe that it should be, wherever possible. Once the necessary information is obtained it will be relatively straight-forward, though sometimes laborious, to calculate how much of a waterbody (in terms of areal proportion) was shallower than the euphotic depth at the time when measurements were taken. This yields a single percentage value indicative of the relative potential for the benthos to contribute to overall primary productivity. This proposed new parameter will be referred to here as the PBI or photic benthos index. The same information can also be used to determine, in percentage of volume terms, how much of the water column (i.e. the limnion) is photic, thereby yielding a PLI or photic limnion index value.

These indices have obvious parallels with the water types discussed in section 4: Waters with low index values will generally be heterotrophic; those with high values will be of the photic-benthic type, and; those with high PLI values and low PBI will be limnetic.

For the purposes of broad-scale regional assessments it should usually be feasible to obtain acceptable PBI and/or PLI estimates by basing calculations on cross-sectional profiles representative of each of the major hydrogeomorphic units contained within a river reach. (River cross-sections similar to those that are required here are already being collected under programs such as Queensland's State of the Rivers). However, for work that is being carried out at sub-regional or local scales, and especially at high priority management and/or research sites, it would be strongly advisable to obtain more detailed bathymetry data, and there are a variety of modern technological aids such as GPS equipped depth sounders that make this an achievable objective. For high resolution work being conducted at sites that support significant assemblages of submergent aquatic macrophytes it would be advisable to calculate an additional index value based on the depth of the underwater canopy. Many modern depth sounders are capable of rapidly providing this information.

The index values provide an ecologically relevant snapshot of the water quality conditions that were prevailing within a waterbody each time it was monitored. However, as discussed in previous sections, the significance of the values obtained cannot be properly assessed without taking antecedences, timing and hydrographic conditions into consideration. Basically it is not the existing light climate *per se*, but rather its context within the longer term light climate regime of the waterbody that determines ecological outcomes. Accordingly the most effective site classification scheme will be the one that is most capable of discriminating key differences in underwater light climate regime, and in order to accomplish this it will be necessary to monitor intensively enough to determine the magnitude and timing of hydrographic, seasonal and/or interannual variations in PBI and PLI values.

None of the monitoring programs that generated the data contained in existing water quality databases (reviewed in the appendix) were sufficiently intensive to allow the detection of key seasonal/hydrographic variations, so the required alterations to monitoring methodology are not confined to the inclusion of new parameters – there is also a need to adopt more appropriate sampling schedules. Notably, since the probability of waterholes being disturbed either by dense congregations of terrestrial animals and/or of pre-flush rain events reaches a maximum during the latter parts of the dry season, it will be essential to ensure that monitoring is carried out at that time of the year. (As will be seen in the Appendix, this critical stage of the dry season is barely represented in most existing datasets). This does not necessarily mean that monitoring needs to be intensified, but rather that steps need to be taken to ensure that key periods and events are covered. (It is noteworthy, however, that very few sites have ever been monitored nearly as intensively as recommended in ANZECC and ARMCANZ (2000) water quality guidelines, so in many cases increased monitoring effort might be advisable regardless of monitoring objectives). The required outcomes could be accomplished by implementing fairly intensive regular interval sampling, but it is feasible and would be far more cost-effective, to adopt targeted risk-based sampling strategies that allow monitoring efforts to be focused on the times and places where key changes are most likely to occur. The considerations discussed in preceding sections provide a basis for determining which sets of hydrological and seasonal conditions to target at different kinds of sites.

Note that although it would be a significant undertaking to collect depth profile data representative of entire river reaches, the work would not necessarily need to be carried out very often, as the channel morphology of many regional rivers only changes substantially during very large floods, and these may have recurrence intervals in the order of decades. It must also be stressed that depth information is not just needed for light climate monitoring; it is in fact an imperative for any ecology-based water quality assessment. For example, as discussed in previous sections, natural instream processes such as evapo-concentration, mixing and stratification, and factors such as water residence time and stormwater detention capacity can have an overwhelming influence on outcomes, and none of these can be determined or understood unless water depths are known.

It is salient to note that existing databases contain many sites that are (somewhat ambitiously) intended to represent the condition of entire river reaches, sometimes hundreds of kilometers long. On some occasions these sites are located in large relatively homogeneous waterbodies that occupy a significant proportion of the reach but on other occasions, especially late in the dry season, they may be situated in small isolated and rapidly retracting pools. Due to the current absence of bathymetry data and associated background information it is not even possible to discriminate between these two contrasting scenarios. In fact none of the geospatial data-sources available for the TRIAP study area provide reliable indications of the permanency of waterholes, so it is currently not even possible to be sure that water and therefore an aquatic ecosystem, is actually present during the dry season. Given the ecological importance of drought refugia, especially in the dry inland catchments, this is a major information deficiency that needs to be addressed.

The development of any truly effective water quality-based classification scheme will also be heavily reliant on the future availability of data indicative of hydrographic variability. However, the aquatic habitats of northern Australia are so spatio-temporally fragmented that it will never be feasible to meaningfully assess regional ecosystems if monitoring is restricted to flow gauging stations; they are far too few in number and since they must be located in places that are conducive to flow measurement, they are seldom representative of more than just a few of the many hydraulic habitats contained within the river system.

Data from gauging stations may provide a basis for predicting hydrological conditions at some nearby sites, but in most cases water quality investigators will need to assess this for themselves. Assessments of this kind do not need to be highly quantitative to be useful; simply knowing for example whether the hydrograph is rising, falling or stable, and/or whether flows are strong, weak or absent, can be a tremendous interpretive aid. In fact the kinds of qualitative information that can be collected by field personnel can actually be more useful than that the quantitative data obtained from flow gauging stations. For example discharge data provide no indications of whether flow conditions at a site were swift and turbulent, or sluggish and calm – a crucial factor that can easily be qualitatively assessed by field personnel, provided of course that they are requested to do so. In the absence of continuous flow records it can sometimes be difficult to determine critical antecedents such as the time since the last swiftflow and/or inflow event. However, since most field personnel must communicate with landholders in order to gain access to properties, it is usually feasible to obtain background information of this sort by interviewing locals.

There are no established methods for collecting or parameterising the kinds of information being discussed here. The ACTFR have been trialing a variety of potential alternatives in the northern GBR catchments and have found that, while different approaches work best in different regions, most methods result in improved data interpretation capabilities. Some of these methods are currently being tested for the first time in the Mitchell River catchment (Queensland) but there are no existing plans to extend this research into other parts of the northern rivers region. These site assessment protocols address most of the water quality drivers discussed in previous sections including stratification and mixing depth, diel periodicity, and the amount and type of instream biomass. Extensive background information of this kind is crucial for understanding the relationship between ecosystem function and ambient water quality.

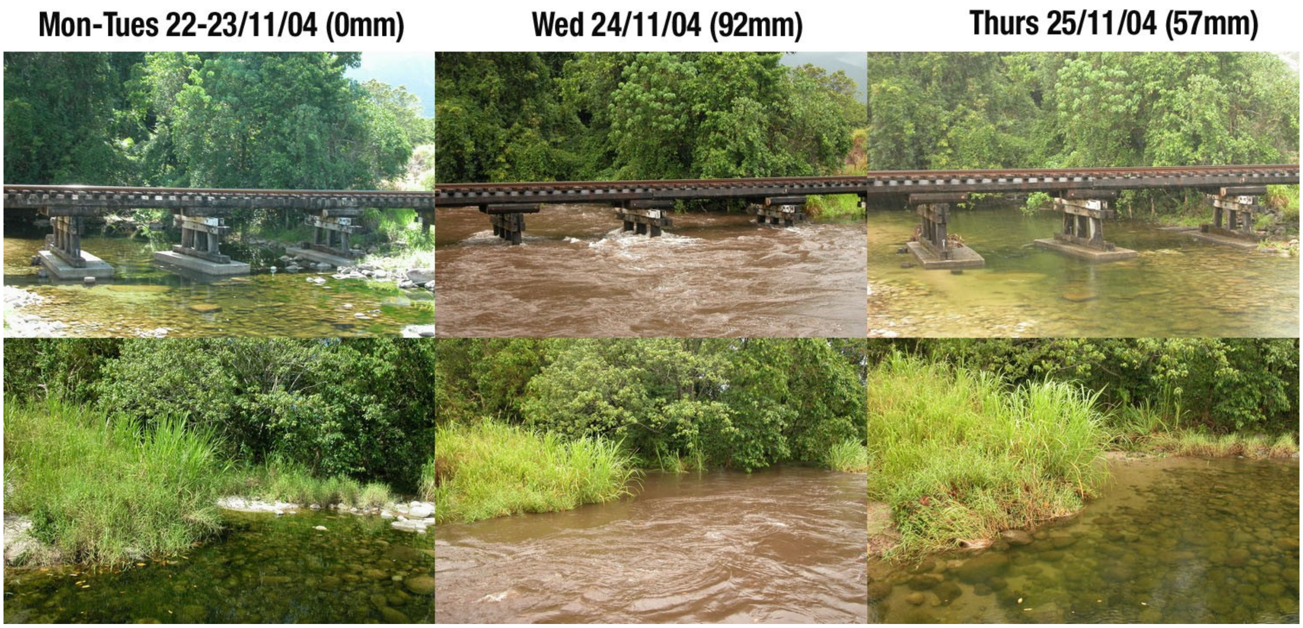
## 5.2 Visual and Photographic Assessment Methods

The optical depth based assessment methods discussed in the subsequent section demand quite intensive field monitoring. If seasonal and interannual regimes are to be determined, this effort will need to be maintained for years (perhaps for decades in the dry inland catchments), hence it would be a major and costly undertaking to gather sufficient data to describe the natural variability of northern river ecosystems. However, one of the advantages of optical depth as a water quality indicator is that it is amenable to simple inexpensive visual and/or photographic assessment techniques, and more importantly, to remote sensing applications.

Photographic methods have great potential as an adjunct to more traditional monitoring techniques, and since no special equipment or training is required, they provide a means by which local community volunteers can participate in professional monitoring programs. Specifically, strategically planned photographic monitoring allows local volunteers to provide scientific personnel with useful information about the kinds of changes that have occurred at monitoring sites between sampling trips. Monitoring of this kind demands close collaboration between participants and must be carefully planned, but if this is done the results can be very effective, especially if volunteers are also provided with a simple data pro forma to record associated details such as daily rainfall and weather conditions.

An experienced limnologist can obtain a surprising amount of useful information from strategically planned photographs. For example, the inferences that can be drawn from the photographs in Figure 17 (and others that were taken at the same time) include, but are by no means limited to: 1) the hydrograph of this site is exceptionally sharp even by local standards; 2) the site is both shallow and optically shallow, except during spates when it very briefly becomes optically deep; 3) instream productivity is primarily algal and there are very few submergent macrophytes; 4) flows are turbulent during spates so re-aeration rates will be high, reducing the risk of DO sags during events; 5) the catchment was green prior to the spate suggesting that there had been recent rain and that the catchment had good groundcover; 6) the benthos was only moderately scoured by flows, so although there was some reduction in benthic algal biomass, the benthos was not reset by this event; 7) baseflow and water transparency were restored within hours of the event, and; 8) the baseflow water level was increased by 25 cm in the aftermath of the event, but this did not result in significant inundation of bank-side vegetation. (Interestingly, in the study from which this example has been taken, there was an adjacent stream that exhibited entirely different temporal variability patterns, remaining turbid and optically deep for the entire wet season).

**Figure 17: Photographic records showing rapid changes in optical depth and flow in Hervey Ck (Russel River catchment, Queensland).**




At detailed study sites it is feasible to survey the streambed and establish reference points that allow many of these characteristics to be determined quite quantitatively, and by conducting flow measurements during a few selected spates it is possible to develop reasonably reliable stage discharge curves, allowing flow rates to be quantified from photographs. This kind of information is a valuable aid for identifying key functional differences between sites, determining when and where to carryout targeted sampling exercises, and interpreting the water quality data that is subsequently obtained. It can also be very useful for prioritising monitoring activities. For example it is obvious from the photographs in Figure 17 that this is not a site that is prone to stratification or the development of hypoxia, so it would be not be worth expending limited resources by carrying out detailed depth profiles or closely examining diel DO periodicity patterns – such work would be better focused on sites that appear likely to need it.

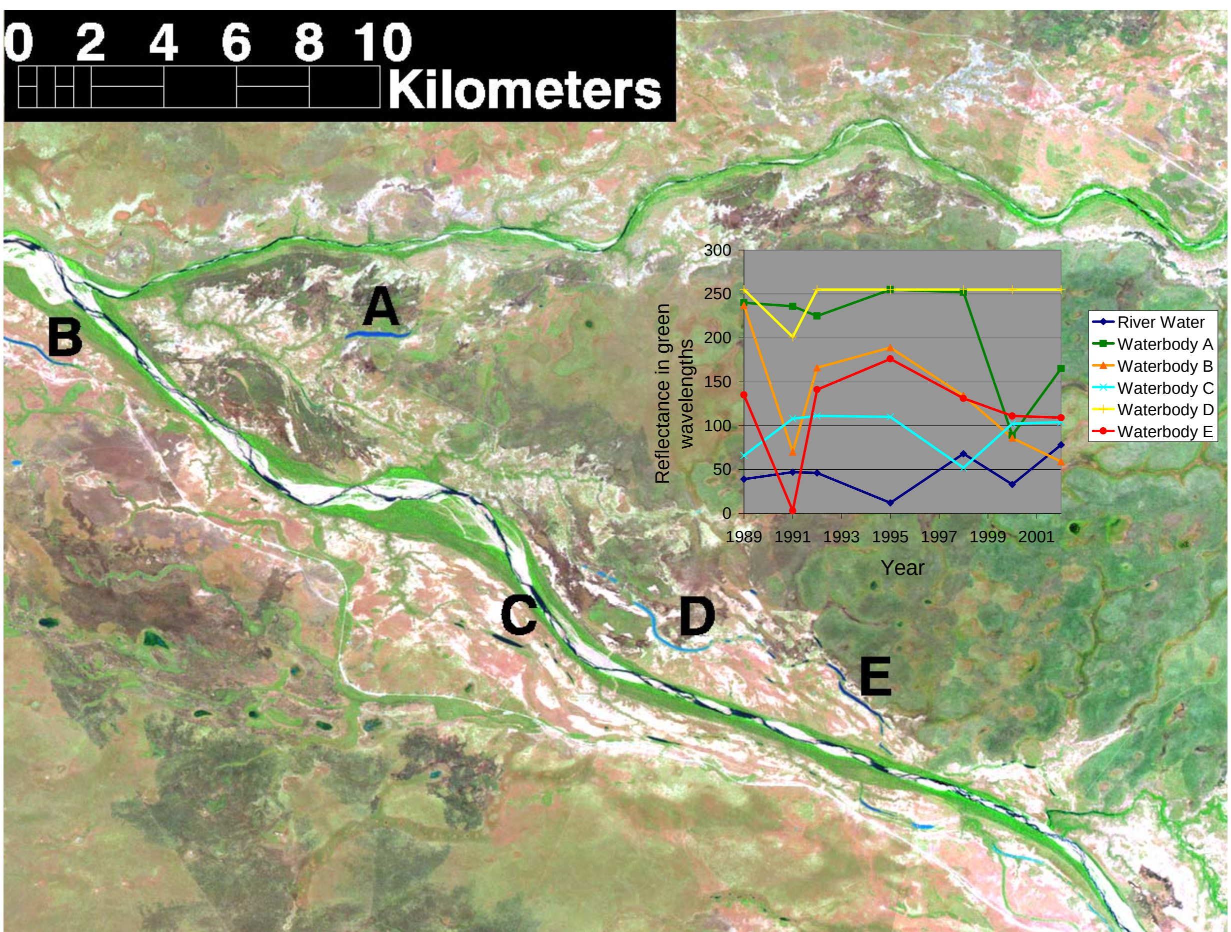
The methods proposed here can substantially enhance monitoring efficiency and are highly recommended, especially when dealing with waterbodies that have been identified as priority sites by managers and/or researchers. However, the diversity, size and remoteness of northern rivers is such that, in the absence of any existing geospatial classification system capable of identifying waterbodies that share key functional traits, there is very limited scope to extrapolate the findings of site-based case studies to the rest of the river system. And as mentioned earlier, the prospects of conducting enough field work to even begin developing a suitable classification scheme in the foreseeable future are not good. Fortunately, however, it may be feasible to rapidly develop a compatible, albeit somewhat coarser, light-climate-based classification scheme by using pre-existing remote sensing imagery to retrospectively analyse historical variations in optical depth.

# 6. REMOTE SENSING AS A WATER QUALITY CLASSIFICATION TOOL

Remote sensing offers three distinct advantages over field sampling based methods; 1) it provides a broad spatial overview of entire drainage systems including off-channel waterbodies and inaccessible reaches, 2) it can make use of existing imagery to examine past variability patterns, and 3) it can also provide supporting information about other important factors such as the permanency and size of waterbodies, the stability of their geomorphology, and their riparian vegetation characteristics.

The first of these can be very important because, as can be seen in Figure 18 (Lymburner *et al*, 2007), spatial variability can be substantial. On this figure optically shallow (relatively clear) waters appear as dark blue and optically deep (turbid) waters appear as light blue. These river reaches tend to run clear on the tail of the hydrograph, and as a result the waters in the main river channel are optically shallow (meaning that they would be expected to report high PLI values and significant PBI values). However, anabranches sometimes stop flowing before stormwaters have been washed away, and when this happens they can remain persistently turbid for the remainder of the year (meaning that they would be expected to report low PLI and PBI values). When this image was taken (in 1992), all of the anabranch waterbodies were in this condition, except for the two sites labeled E and C. However, as can be seen on the inset graph, interannual variability patterns also differ significantly between sites, and this provides an ecologically meaningful basis for classification. Note that although the river channel (clear), site C (reasonably clear) and site D (turbid) varied little over time, other sites fluctuated substantially from year to year. The erratic fluctuations observed at these latter sites, are typical of many regional waterbodies in the drier inland catchments, and demonstrate why it is sometimes necessary to monitor for decades in order to quantify variability patterns. In this case the water contained in the river's baseflow channel was always quite optically shallow, and while this is typical of river reaches that support high baseflows (relative to their channel size), it is by no means characteristic of all rivers in the region.

**Figure 18: A satellite image showing the confluence of the Mitchell and Palmer Rivers (Queensland).**



It is obvious from the above figure that satellite imagery can yield much useful information about the location, number, size, physical complexity, and connectivity of waterbodies, as well as their water permanency, antecedent condition and light climate regimes. As mentioned earlier in this report it can also provide useful insights into the salinity regimes of the tidally influenced waters associated with the lower river reaches. There is clearly great potential to use information of this kind to develop heuristic, ecologically meaningful river reach classification schemes suitable for broad-scale applications. Further research on this topic is urgently needed in order to fully exploit this potential. Specifically there is a need to optimise the interpretation of spectral reflectance signatures, carry out calibrations in the many different kinds of waterbodies and landscapes that occur across the study area, and closely examine the relationships between the apparent optical depths obtained from satellite imagery and estimates obtained on the ground. It will then be necessary to determine the best means of incorporating this information into a classification scheme that is compatible with the field-based site-scale assessment methods discussed in previous sections.

Like most good research, this work will be valuable as much for the questions it poses as it is for the answers it provides. In particular it will provide an excellent basis for determining not only when and where to conduct field investigations, but also precisely what to measure and why. Basically it will allow the formulation of hypotheses that can be tested at relatively low cost either by conducting brief targeted field assessments, or by referring to pre-existing data sources to determine what catchment and stream conditions were like at the time when images were taken. For instance many of the variations plotted on the graph in Figure 18 may be attributed to floods and droughts, and can often be explained by consulting existing hydrological and/or meteorological records. Others are driven by more localised factors, and although there may not be historical data available for the parameters in question, local aquatic systems are sufficiently well understood to be able to identify what field investigations need to be carried in order to determine the source of variation.

As is the case with any broad-scale monitoring technique, remote sensing has some inherent constraints. Landsat provides the best spatial resolution at 25 to 30m, but for analysis purposes this limits its application to waterbodies larger than about 0.2 ha. This means that it can only be used to assess fairly large rivers and tributaries of the kind shown in the above figure. Discrete Landsat images are cost-prohibitive for most environmental monitoring applications, but composite images are freely available. These provide rather disjointed snapshots that are broadly indicative of seasonal conditions, but they do not provide adequate temporal resolution to be able to monitor changes that occur within a season.

The apparent optical depths determined from Landsat images are analogous to those that would be measured in ecological assessments, but they are not the same. Landsat senses the spectral characteristics of the sunlight reflected by bodies of water. If the water is sufficiently transparent and/or shallow to allow the sensors to see the light reflected by the substratum, the waterbody is said to be optically shallow. Conversely, if the water is too deep and/or opaque for the sensors to see the bottom it is said to be optically deep. This differentiation is only possible if the water and the substratum have contrasting spectral reflectance properties, and in most rivers they do – streambeds being composed mainly of sand, gravel and rock, while the suspensoids in the water column comprise mainly clays, colloids and plankton.

There are of course some waterbodies, especially in off-channel wetland systems, where the spectral contrasts between the water and the substratum are weaker and more difficult to detect. This could be the case for example, if the substratum is muddy and the water column becomes turbid due to resuspension of bottom sediments. More significantly, the optical properties of waters and substrata vary substantially over time and space both within and between rivers, so extensive field calibrations will be needed in order to successfully exploit this method.

From an ecological perspective water is optically shallow if it simply allows light to reach the bottom – it does not have to be sufficiently transparent or shallow to allow the light reflected from the bottom to pass through the water column a second time in order to be seen at the surface. There are many waters in this region that would be appear to be optically deep in satellite images because they are too turbid or deep to enable detection of bottom reflectance even though they are optically shallow enough to maintain photic conditions within their benthos. It is possible that discrepancies such as this will eventually be overcome by developing more sophisticated methods of analysing remote sensing data, but that would require extensive research to more closely examine the relationships between spectral reflectance, optical depth and water clarity. Nevertheless, regardless of whether this is accomplished, the relatively coarse semi-quantitative estimates of optical depth that can already be obtained from satellite imagery are still very useful and currently represent the only viable means of obtaining any realistic indications of how light climates vary throughout entire river systems.

Most significantly, Landsat imagery dates back to 1972 (although resolution is better after 1988). It can therefore be used to retrospectively assess historical interannual and seasonal variability patterns. One of the recurrent themes of this report is that the ecological significance of any given water quality condition can vary enormously depending on when it occurred and what the antecedent conditions were like. Basically, in this region of Australia, average and/or instantaneous water quality conditions will never provide an ecologically meaningful basis for assessing or classifying inland waters until the water quality regimes of the rivers are understood. In other words it is imperative to find out how the quality of the water contained within river systems varies over space and time, and analysis of pre-existing satellite imagery provides a unique opportunity to rapidly advance our knowledge in this regard.

Preliminary assessments using methods of this kind have recently been conducted on all of the Queensland Gulf catchments from the Mitchell River west to the Northern Territory border (Lymburner and Burrows 2008a, 2008b). This work entailed using 20 years of dry season imagery to assess water permanency regimes, and 10 years of imagery to evaluate interannual trends in water transparency. The information being generated is already proving to be a valuable aid to regional NRM groups, but further methodological refinements will be needed if the approach is to achieve its full potential.

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# APPENDIX A. ANALYSIS OF EXISTING STATE WATER QUALITY DATASETS

Regional water quality data were obtained from the agencies listed in Table A.1. These comprise the vast majority of publicly accessible data available for the study region. As would be expected, databases from different sources were structured differently, and contained dissimilar parameters, parameter names and units. Data were re-organised, and where necessary recalculated, so that they could be collated into one coherent database for each state. Each of these databases has been structured as similarly as possible, but significant differences remain because different types of data are available in each state.

The available data were originally collected from numerous *ad hoc* projects, none of which were designed to answer the questions being posed in this study, so it was not particularly surprising to find that very few of the data proved to be suitable for our purposes. This report identifies numerous data “deficiencies” and proposes some alternative monitoring approaches that could be applied in order to obtain more ecologically meaningful results in the future. It is important that readers understand that the purpose of this review was to ascertain if any of the existing data could be used as an aid for elucidating the particular water quality processes that are of interest to this study. Data that are not capable of performing this function are said to be deficient, but this assessment should not be taken out of context – it does not imply that the data failed to serve its intended purpose nor is it a criticism of the projects that collected the data.

It must also be stressed that a lot of the data were collected a long time ago, so it would be unrealistic to expect methodologies to be consistent with contemporary best practices. Moreover, even if they were, there would undoubtedly still be some deficiencies because, although many contemporary ecological water quality monitoring practices work well in other parts of the world, they are not nearly so well suited to the highly variable and intermittent freshwater systems of northern and inland Australia. The scientific community has long been aware of this problem and new approaches are constantly being tested, however, reliable and affordable alternatives have proven elusive.

The datasets available to this study do not contain site classifications of any sort, even though they include data from many strikingly different types of sites, including artificial waters (impoundments, drains, pipe outlets), lakes and off-channel wetlands, minor tributaries and distributaries, intermittent waterholes and perennial river reaches. Most sites are inland waters but, based on EC values and mapped locations, some are obviously located in tidally influenced waters and are representative of entirely different ecosystem types. Fundamental differences of these kinds can sometimes be gauged from site names, but that is not a reliable procedure because the datasets contain many nebulous site identifications that have meaning only to the original investigators.

Moreover, monitoring has historically been focused heavily in subcatchments that are subject to unusual human development pressures, such as the irrigation schemes of the Ord and Mitchell/Walsh Rivers, and the uranium mines in the Alligator rivers region. Some of the sites in these areas have been substantially and permanently altered by human intervention, and/or have been strategically selected to allow detection of localised anthropogenic impacts. The records available for these regions undoubtedly also include reference sites and/or pre-impact data, but these are not easily identified without consulting the primary data sources – a time-consuming procedure that falls well outside the scope and resource capabilities of this review. Accordingly, as it stands, any analysis based on pooled data is unlikely to provide reliable indications of natural riverine conditions and/or processes.

Datasets were explored in MS-excel by using a combination of pivot table reports, pivot charts, interactive scatter plots, and customised calculation sheets and Virtual Basic Macros. The exploration process involved the generation of very large numbers of volatile interactive graphical and tabular outputs. Due to space and time constraints it has only been feasible to include a few examples in this report. It should also be noted that the charts shown in the examples are designed for on-screen interactive applications and are not really conducive to the generation of static hard copies. For example there are too many sites on most plots to be able to display identifications on the category axis of hard copies. This is not a problem when viewing the plots on screen because site identifications can be determined by simply resting the cursor on the data series of interest and/or by displaying separate subsets the data at a time. Similarly some of the data shown on hard copies of bar plots are obscured by taller bars, but this is not a problem for onscreen work because the graph can be spun around and viewed from any orientation.

**Table A.1: Data Sources used in this review**

| STATE | CATCHMENT/BASIN | DATA SOURCE | CONTACT |
|---|---|---|---|
| QLD | Archer, Coleman, Ducie, Embley, Flinders, Gilbert, Holroyd, Jardine, Leichhardt, Mitchell, Nicholson, Norman, Staaten, Watson, Wenlock | Natural Resources Mines & Water<br><br>Ph: 0740484872<br>Fax: 0740922366<br>www.nrm.qld.gov.au/watershed/index.html | Vince Manley<br>*Senior Hydrographer*<br><br>Geoff Pocock<br>*Senior Hydrographer* |
| WA | Drysdale, Fitzroy, Isdell, Keep, King, Edward, Lennard, Lower Ord, Pentecost, Upper Ord | Department of Water<br>Water Information Branch<br>Ph: (08) 63647469<br>Fax: (08) 94264821<br>waterinfo@water.wa.gov.au<br><br>Email: muriel.cordemans@water.wa.gov.au | Muriel Cordemans<br>*Data Management Officer* |
| WA | Ord River | Environmental Research Institute of Supervising Scientist<br>Ph : +61 8 89 201160<br>Fax: +61 8 89 201195<br><br>Email: chris.humphrey@deh.gov.au<br>Email: Eugene.Carew@doir.wa.gov.au | Chris Humphries<br>*ERISS*<br><br>Eugene Carew |
| WA | Cape Leveque, Drysdale, Fitzroy, Isdell, King Edward, Lennard, Ord, Pentecost, Prince Regent | Department of the Environment and Heritage<br>PO Box 461<br>Darwin NT 0801<br>Ph +61 8 8920 1153 Fax +61 8 8920 1199<br><br>Email: John.Lowry@deh.gov.au | John Lowry<br>*ERISS* |
| NT | Adelaide, Blyth, Calvert, Daly, East Alligator, Finniss, Fitzmaurice, Glyde, Goomadeer, Goyder, Lantram, Limmen Bight, Liverpool, Mary, McArthur, Robinson, Roper, South Alligator, Victoria, West Alligator | Department of Natural Resources, Environment and the Arts<br>Environment Protection Agency Program<br>PO BOX 30<br>PALMERSTON NT 0831<br>Ph: (08) 8999 4406<br>Fax: (08) 8999 4403<br><br>Email: julia.fortune@nt.gov.au | Julia Fortune<br>*Senior Scientist*<br>*Aquatic Health Unit* |
| NT | Adelaide, Daly, East Alligator, Mary, McArthur, Robinson, Roper, South Alligator, Victoria | Department of the Environment and Heritage<br>PO Box 461<br>Darwin NT 0801<br>Ph +61 8 8920 1153 Fax +61 8 8920 1199<br>Email: John.Lowry@deh.gov.au | John Lowry<br>*ERISS* |

## A.1 Queensland

The Queensland northern rivers water quality dataset collated for this project comprises 5467 records that contain data for at least one water quality parameter. The data were collected over more than 4 decades between 1961 and 2005 from 174 sites, covering 15 river basins; the most well-represented being the Mitchell (89 sites) and Gilbert (29 sites). Spatial coverage of the remaining 13 basins is sparse (56 sites in total) and catchments such as the Flinders, Embley and Watson Rivers are represented by only one or two sites. Temporal coverage is both sparse and heterogeneous.

As discussed in section 4, turbidity and SPM are the parameters of greatest potential value to this study. However, exploratory analysis showed that temporal coverages at most sites were too poor to provide real-time estimates of hydrographic and/or seasonal dynamics. Figure A.1 for example plots the numbers of turbidity and/or SPM values collected annually at each site. Each horizontal row of bars on this plot represents a monitoring site (unfortunately, as explained earlier, there are too many sites to be able to display legible labels on the category axis). During analysis the time scale was expanded to view monthly rather than annual sample distributions. This showed that most of the high annual counts could actually be attributed to brief intensive monitoring events with durations of less than one or two months. Moreover, almost all of the sites with sufficient data to potentially be useful for elucidating the key natural water quality processes discussed in the main text of this report, were located in the Mitchell River Catchment.

During the course of this study the large Mitchell River dataset was reviewed in some detail in connection with other projects (Butler and Burrows 2005). That review found that most of the data were inadequate to enable detection of the kinds of spatio-temporal variability patterns that are of particular interest here. However, the database did contain some sites in the upper Walsh and Mitchell Rivers where turbidity/SPM (and EC) had been monitored intensively enough during the wet season to warrant closer examination (these are the very large bars on figure A.1). These data were collected at regular intervals during storm events, and although some key stages of the hydrograph were missed, they still adequately demonstrated some key elements of the dynamics of fluvial sediment transportation in this catchment (Butler and Burrows 2007a). For example Figure 13 in section 2.3 was derived from these data.

The majority of samples (5375 out of 5467) have been analysed for EC. The data indicate that the EC levels of all freshwater sites (i.e. those not influenced by seawater) are very low to moderate, rarely exceeding 800 µS $cm^{-1}$. Available data indicate that most freshwater organisms can tolerate salinity levels substantially higher than this; toxicological effects not normally being observed until EC levels significantly exceed 1500 µS $cm^{-1}$ (see main text for references). As noted previously, there may be some non-toxicological effects, especially when EC concentrations are very low. For example Horrigan *et al* (2005) analysed the Queensland AusRivAS dataset (including the northern rivers data) and found statistically significant relationships between EC and macroinvertebrate community composition. However, effects were greatest at sites with EC levels less than 130 µS $cm^{-1}$, and diminished rapidly with increasing EC, becoming insignificant at about 800 µS $cm^{-1}$. The observed changes were quite subtle (the number of taxa for example was unaffected) and they provide no indication of potential for major functional transitions of the kind that are being focused on in the current study.

EC data could still potentially be a useful indicator of the dynamics of the natural processes that drive these ecological transitions, but the Mitchell River is the only system where sites have been sampled intensively enough to achieve the required temporal/hydrographic resolution. Most of these sites lie within and downstream of the irrigation areas located in the upper Mitchell and Walsh River catchments, so the available data are not necessarily indicative of natural variability patterns. Butler and Burrows (2006a) conducted a detailed examination of the upper Walsh River data and concluded that, although there was no evidence of salinisation effects from the agricultural area, there was unequivocal evidence that the hydrological modifications caused by river regulation and/or distribution of irrigation water, had substantially altered the river's seasonal EC variability patterns. Specifically, due to flow supplementation, increased water depths and/or increased detention of low salinity wet season runoff, EC values in the river were no longer able to rise substantially during the dry season, as they would have done when the river was in its natural state. There were sufficient data available from unsupplemented tributary streams, to be able to confirm that seasonal increases in conductivity could be quite significant, especially at shallow sites, and that the observed increases were consistent with predicted natural evapo-concentration rates.

Data relating to alkalinity, hardness and other major ionic constituents are not quite as extensive as the EC records, but they are available for the majority of samples at most sites. McNeil and Cox (2000) have analysed these data to determine the relationships between EC and total dissolved ion concentrations, McNeil *et al* (2005) were able to use the data to classify Queensland's inland waters into nine major types based on their ionic composition. There are no grounds to expect these classifications to correlate to major differences in the functional ecology of sites – the identified classes have ion concentrations that are not high enough to be toxic to most freshwater organisms, and certainly not to entire functional groups. However, several of the classes have EC ranges which, based on the findings of Horrigan *et al* (2005), are likely to support distinctive macroinvertebrate community types. Two classes contained very low proportions of calcium and magnesium ions, suggesting potential for conditions to become nutrient limiting for some faunal species. Accordingly, there may be some future potential to include ionic composition related parameters as sub-classes or class-modifiers in a broader geo-spatial ecological classification scheme.

However, in order to accomplish this it will be necessary to properly discriminate between spatial and temporal variations in ionic composition. It may also be necessary to develop parameterisation methods capable of identifying and expressing key differences in temporal variability patterns before attempting spatial classifications. Basically, ecologically meaningful classifications will be obtained not by analysing how water quality conditions vary over space, but rather by determining how temporal water quality variability patterns vary over space. If existing datasets were representative of all stages of the hydrograph and/or times of the year, it might be feasible to accomplish this empirically by simply employing multivariate statistical analysis methods. There is little doubt that this would lead to the identification of some statistically distinctive water quality types. However, since the effects of water quality changes are non-linear, sometimes subject to thresholds and are dependent upon antecedents, an exceptionally sophisticated statistical model would be needed to be able to identify classes that have any known ecological relevance. In practice this is a moot point because the spatial and temporal coverages of existing data are completely inadequate and there are no prospects of this situation improving in the foreseeable future.

The between-site differences in sampling frequency and timing in the Queensland dataset are currently so large that, although it might be possible to identify water types with characteristic ionic signatures, it is not possible to determine if these are characteristic of a site or of a certain hydrographic or seasonal condition. Many sites have been intensively monitored only during storm events and as a consequence they report very low EC values and an ionic composition equivalent to extremely dilute seawater. There are a number of perennial site that have been monitored mainly under baseflow conditions during the middle of the dry season and these predictably report moderate EC values with increased proportions of calcium, magnesium and alkalinity. To date however, the substantially altered reaches in the upper Walsh and Mitchell rivers are the only sites that have been monitored consistently enough to obtain any realistic estimates of natural seasonal variability patterns, and even in these cases the data only cover a few years and do not adequately indicate interannual variability. In particular there is a paucity of data indicative of the late dry season (the time of the year when ion concentrations usually reach an annual maximum) and droughts (the years when ion concentrations are most likely to reach long term maxima).

Data for all other water quality parameters are extremely sparse. Comprehensive nutrient speciation data (395 samples with concentration values for particulate and dissolved forms of nitrogen and phosphorus) were collected for only a few years and only at 33 sites located in the reaches and tributaries of the upper Mitchell and Walsh Rivers. These sites are mostly located in hydrologically altered streams within or downstream of intensive agricultural and/or urban developments. They do not therefore provide a basis for assessing natural variability patterns.

The dataset contains fields for two types of nitrate analyses – one where values are reported in mg of $NO_3$/L (the unit of choice for potability assessments and ionic composition studies) and a second where values are reported in µg N/L (the unit of choice for ecological studies). The $NO_3$ results are highly suspect (most of the analyses were conducted on unpreserved samples and mean concentrations are too high to be believable) and in most cases where other nutrient estimates were available, there was general disagreement between results. In several cases the reported $NO_3$ value was actually greater than the total N value, suggesting that incorrect sample preservation may not have been the only source of error (mg $NO_3$/L values were converted to µg N/L in order to make this comparison). Accordingly the $NO_3$ data have been rejected. The results contained in the other nitrate field appear to be valid and have been retained. Concentration values and nutrient species ratios are both broadly consistent with normal expectations for these sites.

A further 477 samples from 37 additional sites have data available for one or more nutrient. However, most of the data were collected during wet season storm events and are not indicative of normal ambient conditions. Nutrient concentrations fluctuate so substantially, both during and between events, that it is not possible to obtain meaningful estimates of event concentrations without monitoring intensively through the hydrograph. This has been done for only a few events and only at sites in the developed areas in the upper Mitchell catchment.

The bulk of the remaining data relate to trace constituents including some metals and/or anthropogenic contaminants, although there are some data available for water colour. Spatial and temporal coverages are far too poor for these to be useful to the current study. Moreover, given the enormous number of disused and/or abandoned mine sites in the region it would be a major undertaking to determine if trace metals values are indicative of natural levels (there are in excess of 4000 historical mine sites in the upper Mitchell catchment alone).

**Figure A.1:** **Queensland northern rivers dataset - Counts of cases where turbidity and/or SPM values are available, clustered by year and site. (Each horizontal row of bars denotes a site).**

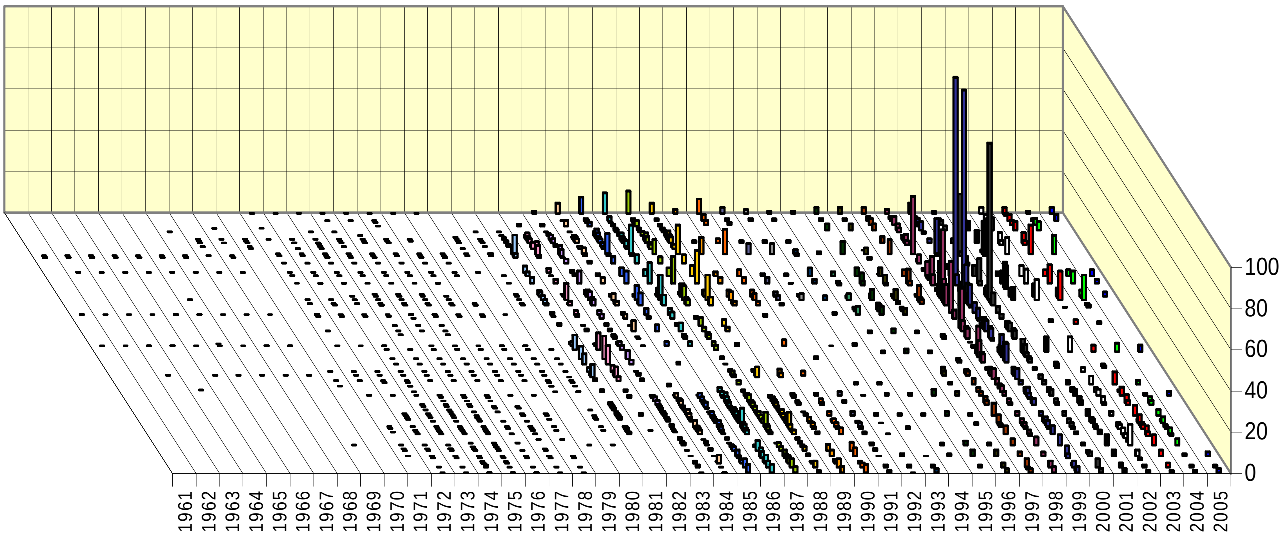
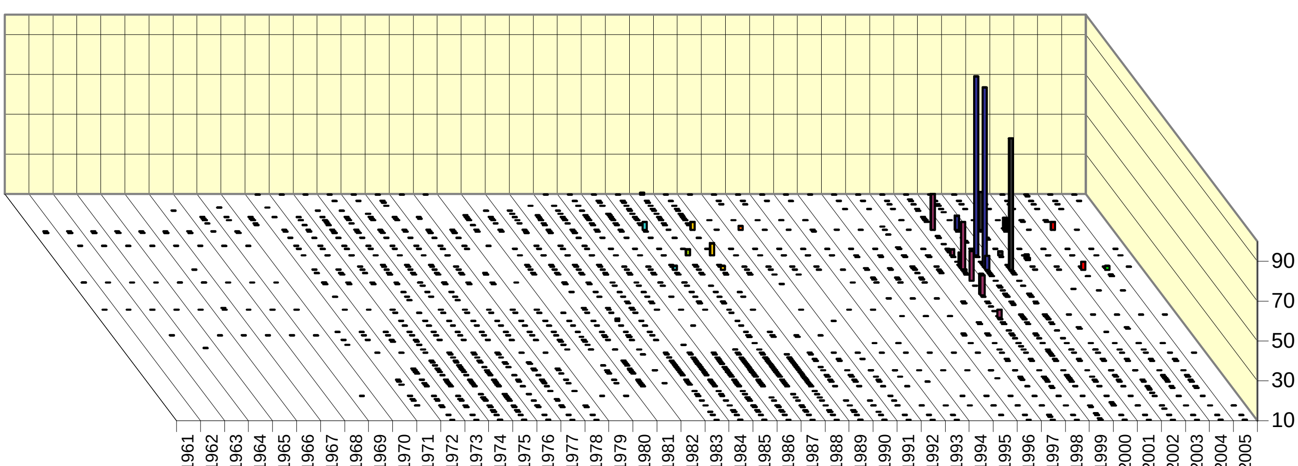



**Figure A.2 :** **Queensland northern rivers dataset - Counts of cases where more than 10 turbidity and/or SPM values are available.**



## A.2 Western Australia

In terms of total numbers of records and fields, the WA dataset is larger than Queensland's. However, it is much less internally consistent and contains considerable numbers of records that are almost blank. In fact only 9535 of the 21241 records actually contain any water quality data that could be relevant to this study. Out of the remaining 11706 records, 7796 (more than one third of the total dataset) only contain storage levels for Lake Argyle. Most of the remaining records relate to physical data such as stream heights and/or flow rates. Some samples from the lower Ord were analysed for a range of pesticide residues and/or algal species compositions, but the WQ parameters of interest to this study were not analysed.

Data were collected from 254 sites between 1949 and 2006. However, only 110 sites have been sampled on more than 6 occasions (which would be the absolute minimum number of samples required to have any realistic chance of discerning any seasonal variability patterns, and then only if all samples were collected in the same year). Spatial coverage is extremely patchy, 81of these sites being located in the Ord River catchment and 17 in the Fitzroy (leaving only 12 sites to cover the remainder of the study area).

Turbidity and/or SPM values are available in 4565 cases, but only 71 sites have been sampled for these parameters on more than 6 occasions and only 2942 of these records contain any information indicative of hydrographic conditions (stream discharge, stage and/or stream height) at the time of sampling. Some of these relate to sites with names indicative of artificial waters (e.g. drains, pipe outlets, spillways and dams) and have been excluded from subsequent analyses. There is no means of being certain about the status of the remaining sites without consulting the data providers, but they are assumed to be located in natural, though not necessarily undisturbed, waterbodies. This leaves 2111 records representative of the 29 (presumably) natural river sites that have been sampled on more than 6 occasions.

The sparseness and heterogeneity of the data are illustrated in figures A.3 and A.4, which display counts of cases for which both turbidity/SPM and flow records (of some kind) were available. Both figures encompass all 254 monitoring sites. Figure A.3 displays the first 30 years of records and shows that regular collection of turbidity/SPM data did not commence until after 1976. Figure A.4 shows the period from 1977 to 1996, which encompasses more than 95% of the available records. Figure A.5 displays only "natural" sites that have been sampled on more than 6 occasions. It can be seen that sampling was most intensive during the late seventies and mid-eighties, and was heavily focused on just a few sites in the Ord. These few intensive programs generated large numbers of samples but as can be seen in Figure A.6, they were of very limited duration and confined almost exclusively to mid wet season months. There are still some sites that have been sampled a number of times almost every year, however, as Figure A.6 illustrates, there are no sites where collection of both turbidity and flow data was continued over the course of an entire year or even a season. The extreme paucity of mid to late dry season data is particularly noticeable.

Obviously there are insufficient data to be able to evaluate the pseudo-seasonal water clarity regimes of any individual site in any given year. From a purely mathematical perspective there may be some sites with enough data to be able to obtain first-order estimates of long term average seasonal patterns. However, there are sound scientific grounds to doubt the effectiveness of such an approach. Data cannot validly be stratified simply according to the calendar, especially in the drier inland catchment areas, because the major rain events that drive variability do not reliably arrive in the same months each year – October and December for example, can be the times when sediment loading is at either its annual maximum or minimum, depending on whether first-flush rainfall has occurred yet. The data really need to be stratified according to flow "state" but this cannot be accomplished by simply using instantaneous flow rate measurements; in order to obtain meaningful results the stage of the hydrograph and antecedent flow conditions must also be taken into consideration. For example, depending on whether it occurs on the rising stage of a first-flush event, the rising stage of a follow-up event, the falling stage of a small event or the tail of a very large event, the turbidity levels associated with a given flow rate can be enormous, moderately elevated, quite moderate or very low, respectively. The data available for most monitoring sites are not adequate to assess flow conditions in these terms and in the few cases where they are, the required information is not contained in the water quality databases and must be obtained from other sources. Moreover, although these processes are understood in qualitative terms, there are no proven methods of quantitatively defining flow states, and no established parameterisation schemes to facilitate the incorporation of flow status information into water quality databases. These are research topics that deserve urgent attention.

**Figure A.3:** **Western Australian northern rivers dataset - Counts of cases where turbidity/SPM values and stage/discharge/height records are available, clustered by year and site. Each vertical row of bars on the horizontal plane represents one of 254 sites. Only values collected between 1949 and 1983 are shown in this view.**

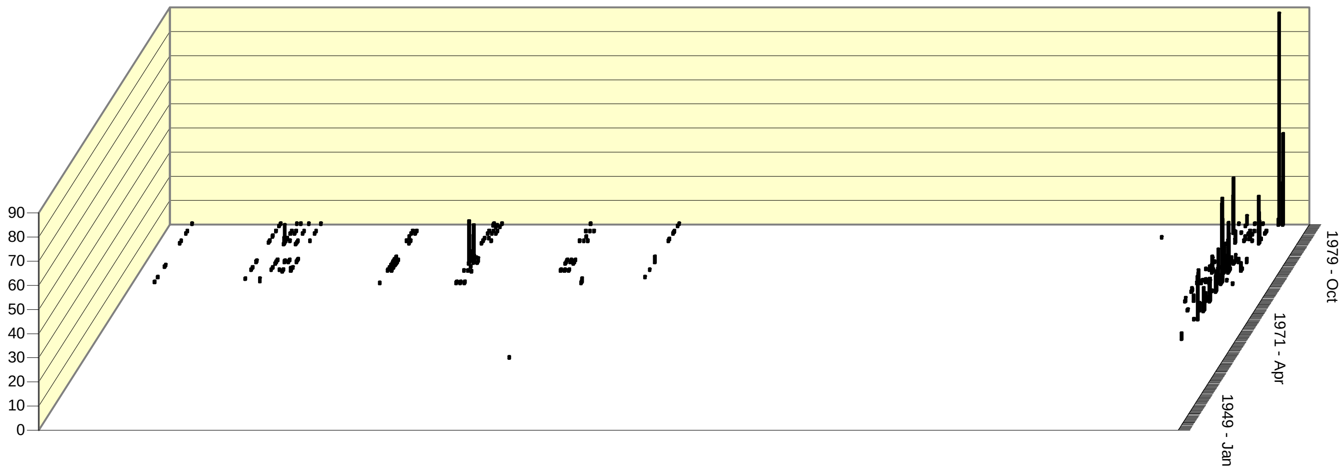



**Figure A.4:** **A rescaled version of Figure A.3 showing only the most intensive sampling period – 1977 to 1996.**

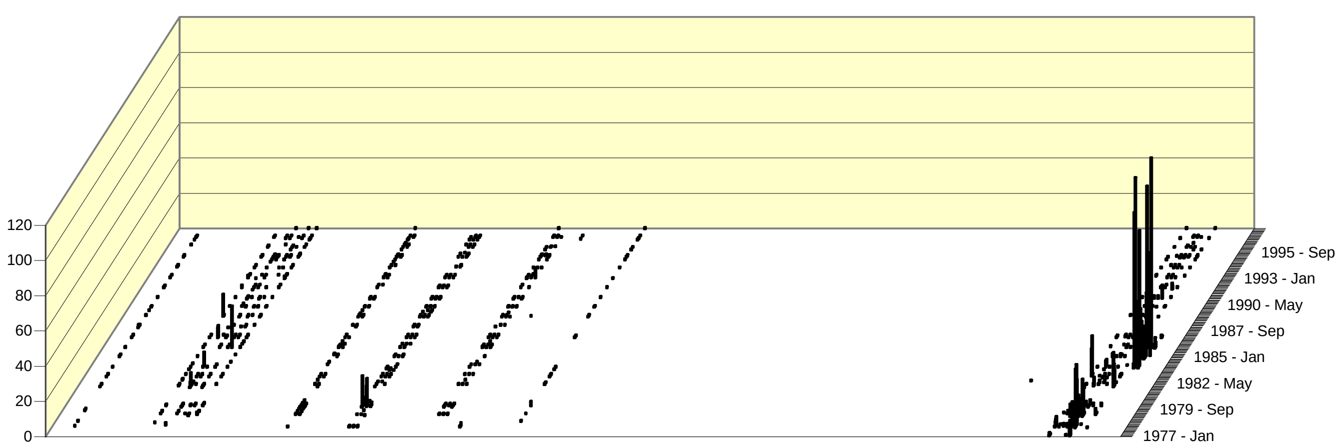


**Figure A.5:** **Western Australian northern rivers dataset - Counts of cases where turbidity/SPM values and stage/discharge/height records are available, clustered by year and site. Only sites that are presumed to be natural and which have been sampled on at least 6 occasions are shown.**

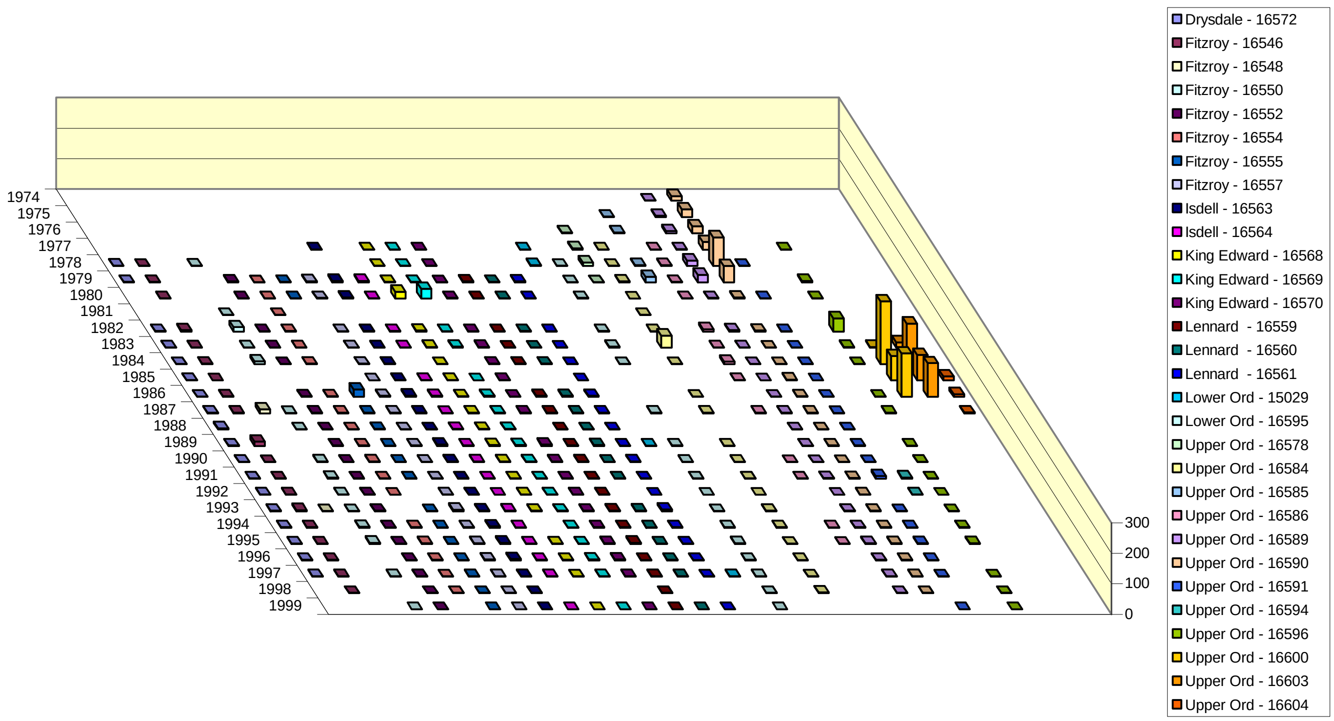
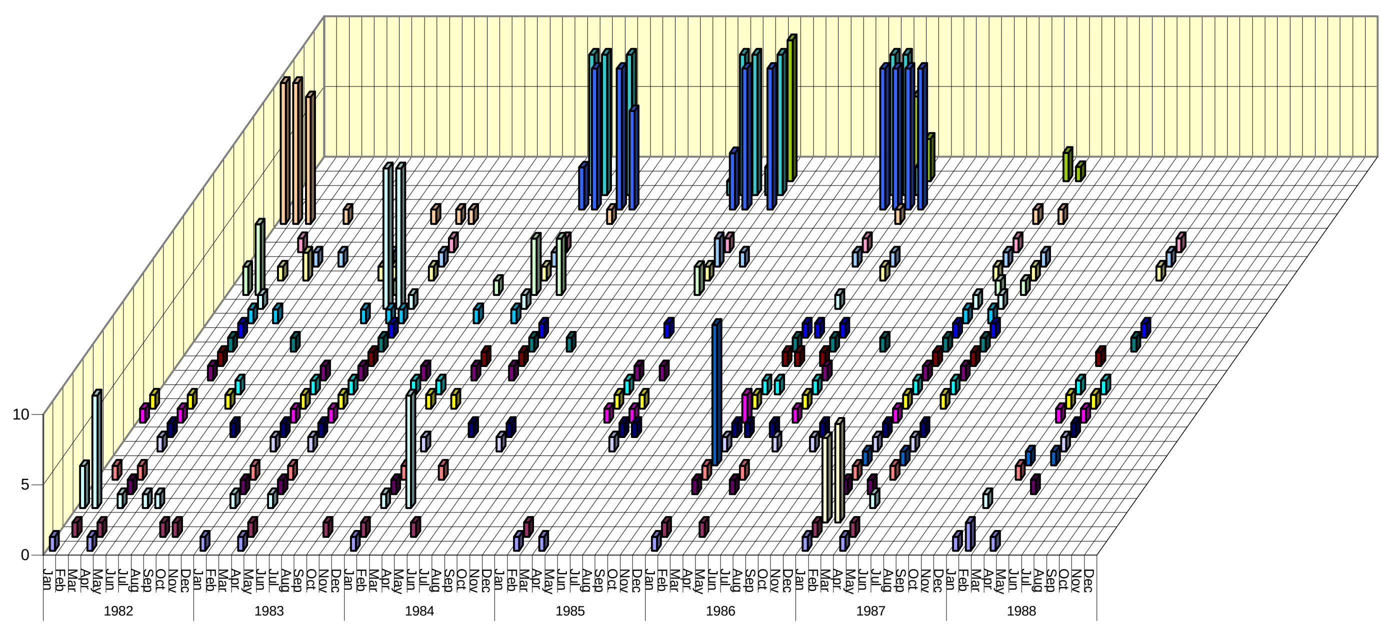



**Figure A.6:** **Western Australian northern rivers dataset – A version of Figure A.5 that has been re-scaled to show monthly time intervals over the most intensive monitoring period. The value axis has been expanded to display counts of less than 10. Bars representative of higher counts are truncated.**



Figures A.7 and A.8 allow qualitative visualization of the available turbidity/SPM data. These plots display turbidity values for all cases where they were available, but employ surrogate SPM values in cases where turbidity values were unavailable. In our personal experience, turbidity to SPM ratios generally range from about 0.6 to 1.7 whenever SPM levels are above about 50 mg/L, but become much more variable at lower concentrations. Accordingly, although as discussed previously, SPM values are not always a reliable quantitative predictor of turbidity, they can provide useful qualitative estimates.

Research in the GBR catchments (see for example Fentie et al 2006, Furnas 2003, McKergow *et al* 2005, Bainbridge *et al* 2006) indicates that the turbidity concentrations in inland rivers of the wet-dry tropics commonly rise to levels in the order of 1000 to 10000 NTU during floods, values depending on the size of the river and its flow state. These extreme levels are seldom maintained for long once flow rates fall, but there are some sites that maintain turbidities in excess of several hundred NTU for the remainder of the wet season. Many of these run clear early in the dry season, but some remain turbid for the entire year. As can be seen in Figure A.7, values indicative of these high wet season event concentrations are confined to just a few sites on the Ord River which were monitored intensively during wet seasons in the mid eighties. The significance of these higher values could not be assessed without consulting the investigators who carried out this work, but the reported concentrations are within the ranges that are normally encountered during flood events in large rivers. The absence of high values for other sites is particularly noticeable, but due to a lack of data indicative of the rising limb of major flood hydrographs, it is not possible to ascertain whether or not extreme turbidity levels occur at these sites during events. The results do suggest that few of them remain highly turbid between flow events, however, as can be seen in the expanded view in Figure A.8, the turbidity levels at most sites are still quite significantly higher during the wet season than they are in the dry. These findings are not inconsistent with the expected patterns outlined in section 2.3, however, the available data are far too limited to be able to detect the kinds of between-site differences in temporal/hydrographic variability patterns that have been hypothesized. Further analysis of these data would be a major and costly undertaking, and is unlikely yield any substantial improvements in the understanding of these river systems.

The WA dataset contains 4386 EC values and 1798 alkalinity values, but significantly less data are available for other ionic constituents – for example only 584 samples have been analysed for all major cations. EC data available from some sources were reported in $\mu S\ m^{-1}$ and had to be divided by 100 in order to convert them to the more conventional $\mu S\ cm^{-1}$ units. Only 9 of the 254 sites for which data are available reported median EC concentrations above 2000 $\mu S\ cm^{-1}$, 7 of these being located in the lower Ord. All but one of these were actually just individual spot measurements (i.e. there was only one EC value available for each site), so there is only one site in the dataset that consistently reported EC values (median 7150 $\mu S\ cm^{-1}$) indicative of brackish water. There are only 82 sites where EC has been determined on more than six occasions, and apart from the one brackish water site in the lower Ord, the highest median EC value reported was only 842 $\mu S\ cm^{-1}$ (maximum 1346 $\mu S\ cm^{-1}$). Given the severe under-representation of late dry season conditions in this dataset it is highly likely that these figures significantly underestimate the salinity ranges of many sites. Nevertheless, the results still suggest that EC is unlikely to be a primary driver of ecosystem function at most of the inland sites that have been monitored to date.

Nutrient speciation has been assessed on more than six occasions at only 26 sites (1182 values). All of these are located within or downstream of the Ord irrigation area and are not necessarily indicative of natural conditions.

**Figure A.7:** **Western Australian northern rivers dataset – Average monthly turbidity (NTU) and/or SPM (mg/L) for the samples shown in Figure A.6. Turbidity values are used in cases where data are available for both parameters.**

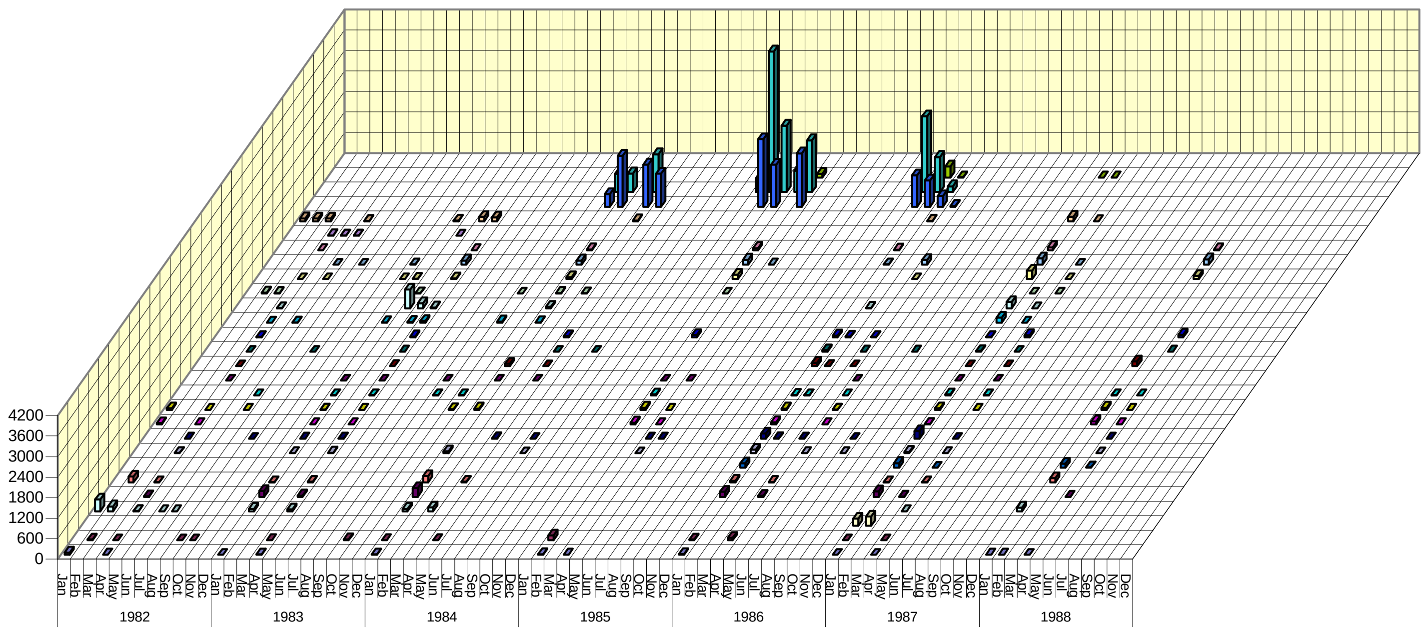
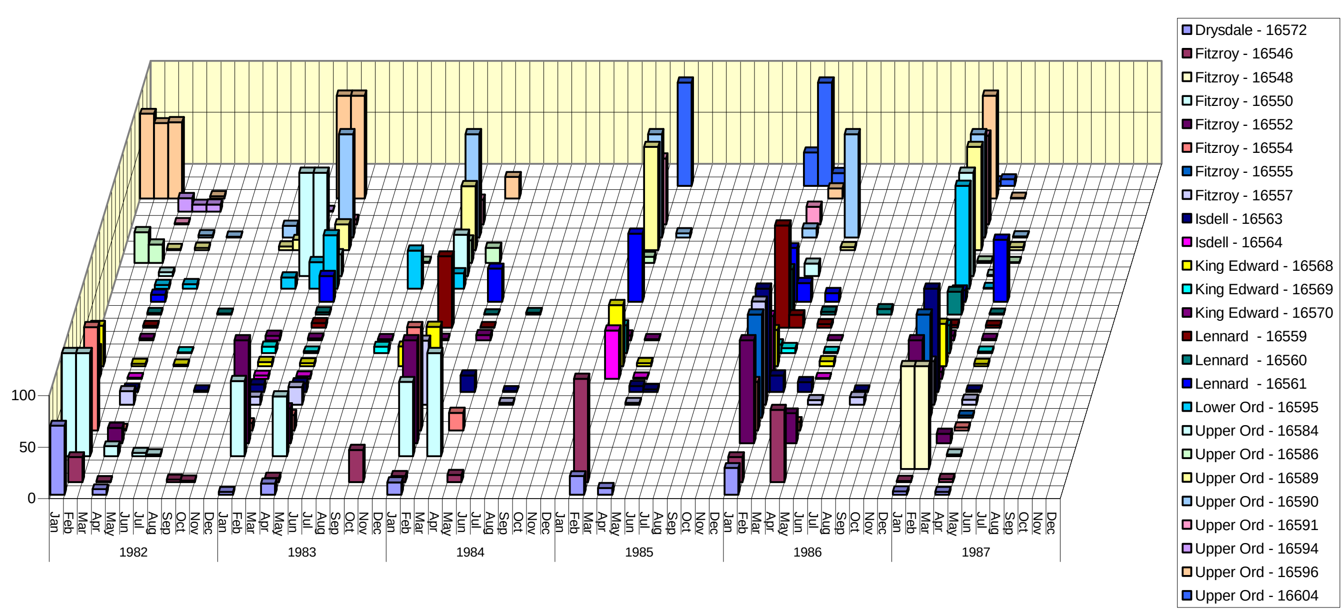



**Figure A.8:** **Western Australian northern rivers dataset – A re-scaled view of Figure A.7. Bars indicative of values greater than 100 (NTU/mg/L) are truncated. The three Ord sites that reported mostly very high values in Figure A.7 are not visible in this view.**



## A.3 Northern Territory

The main purpose of this data review was to determine if existing water quality data could be used for the empirical development an ecologically relevant classification scheme that could be applied across northern Australia. By the time the Northern Territory data were reviewed it had already been established that the Queensland and Western Australian datasets were inadequate for this purpose. Accordingly, the NT data have not been analysed quite as thoroughly as the others.

This is the largest of the three datasets, comprising 26060 records collected from 1611 sites over fifty years. However, it contains large amounts of data for parameters such as metals, which are not of interest to this study, and it also provides very uneven spatio-temporal coverage, especially for the parameters that are being focused on here. For example only 387 of the 1611 sites have been monitored on more than six occasions, and only 171 of those have been sampled for turbidity/SPM more than six times. Moreover, 103 of the latter sites are located in the East or South Alligator River catchments, and these account for 7994 of the 9699 available turbidity/SPM records. This imbalance can be visualised in Figures A.9 and A.10. Notably the Daly River catchment is represented by only two sites – G8140040 on the Daly River (74 results), and G8140063 on the Douglas River (only 8 results).

The dataset also contains relatively few data indicative of flow conditions; for example, as can be seen in Figure A.11, only a small proportion (895 out of 9699) of the turbidity/SPM values displayed in Figure A.9 are supported by discharge or water level measurements of any kind. The sample counts shown in Figure A.11 are still reasonable, but as can be seen in Figure A.12, temporal distributions are very uneven, the majority of sampling having been carried out in the mid to late wet season. As was the case with the WA data, coverage of the late dry season months is particularly poor.

The summary table for this dataset contains 177 cases where median EC values exceeded 2000 µS $cm^{-1}$, suggesting high potential for ecological effects. However, the majority of these sites were only ever monitored on one or two occasions, and there are only 11 instances where elevated medians were reported at sites that had been sampled more than six times. One of these was a marine site (Darwin Harbour), one was identified as a river mouth, one was a dam, six were located at mine sites and the status of the other two is unknown. There are 311 other sites where EC has been determined more than six times. Median values for these sites were generally moderate (averaging 245 µS $cm^{-1}$) but the reported values are somewhat more variable than the EC values in the other State datasets, ranging from 1 µS $cm^{-1}$ (not a misprint) and 1900 µS $cm^{-1}$. It is highly likely that this is due to the inclusion of a greater number of near-coastal, off-channel and/or disturbed sites. However, this would be a difficult hypothesis to check because, as is the case with the other datasets, sites are not classified and in many cases they are identified using quite nebulous site names. Descriptive comments have been provided for some samples but these are not standarised so it is not possible to employ them in automated analyses.

**Figure A.9:** **NT dataset – counts of samples analysed for turbidity and/or SPM at sites that have been sampled on more than 6 occasions.**

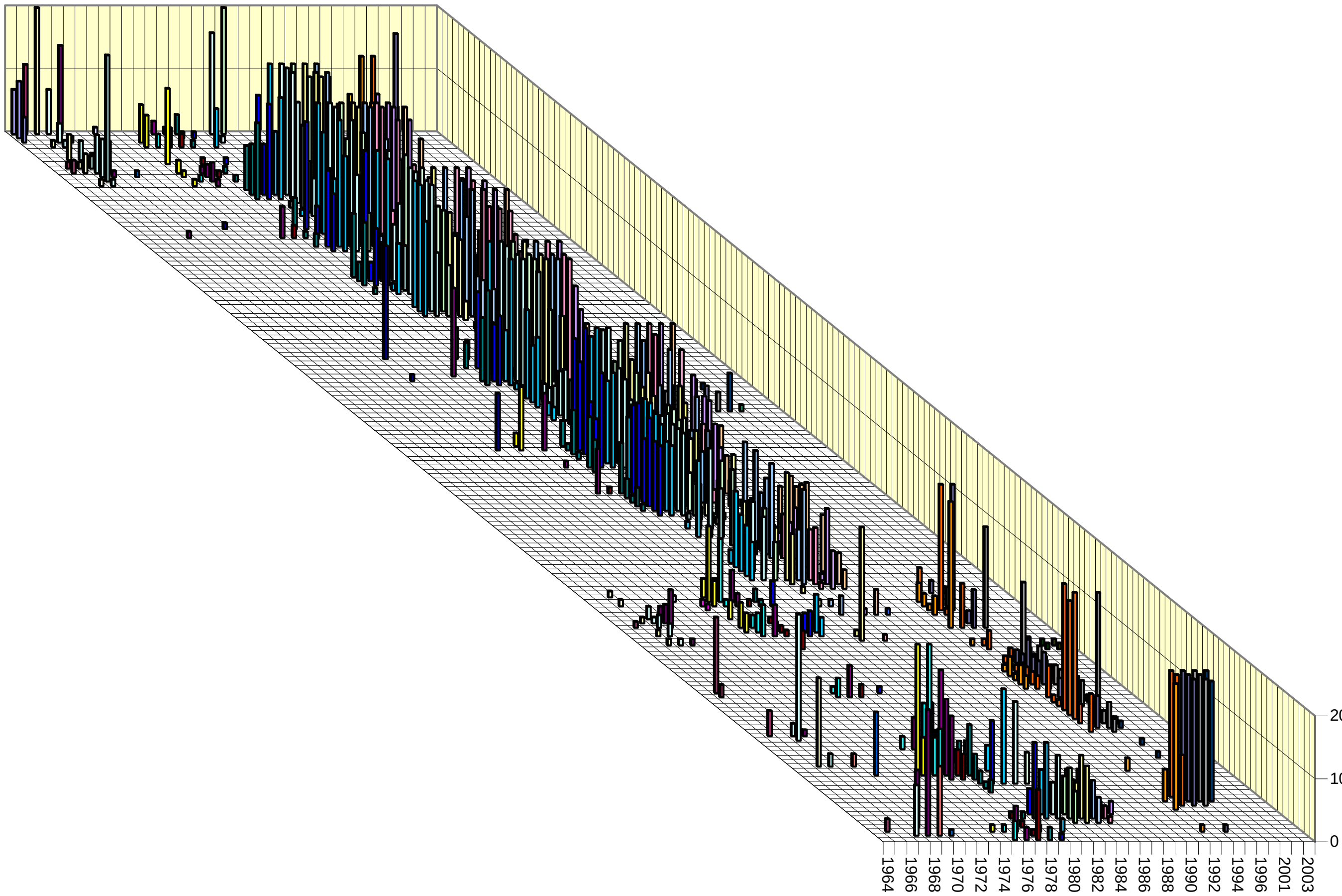



**Figure A.10:** **NT dataset – Figure A.9 with East and South Alligator River Catchments hidden.**

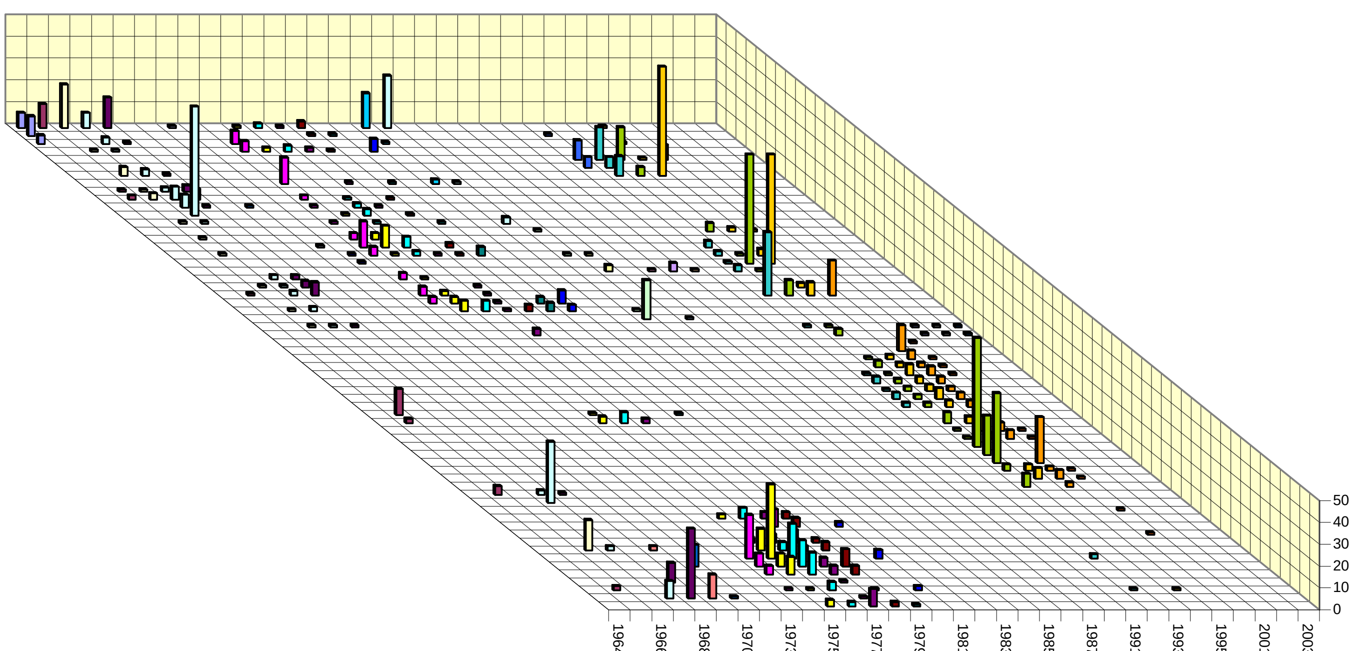


**Figure A.11: NT northern rivers dataset – Counts of cases where turbidity/SPM values and stage/discharge/height records are available for sites that have been sampled on more than 6 occasions.**

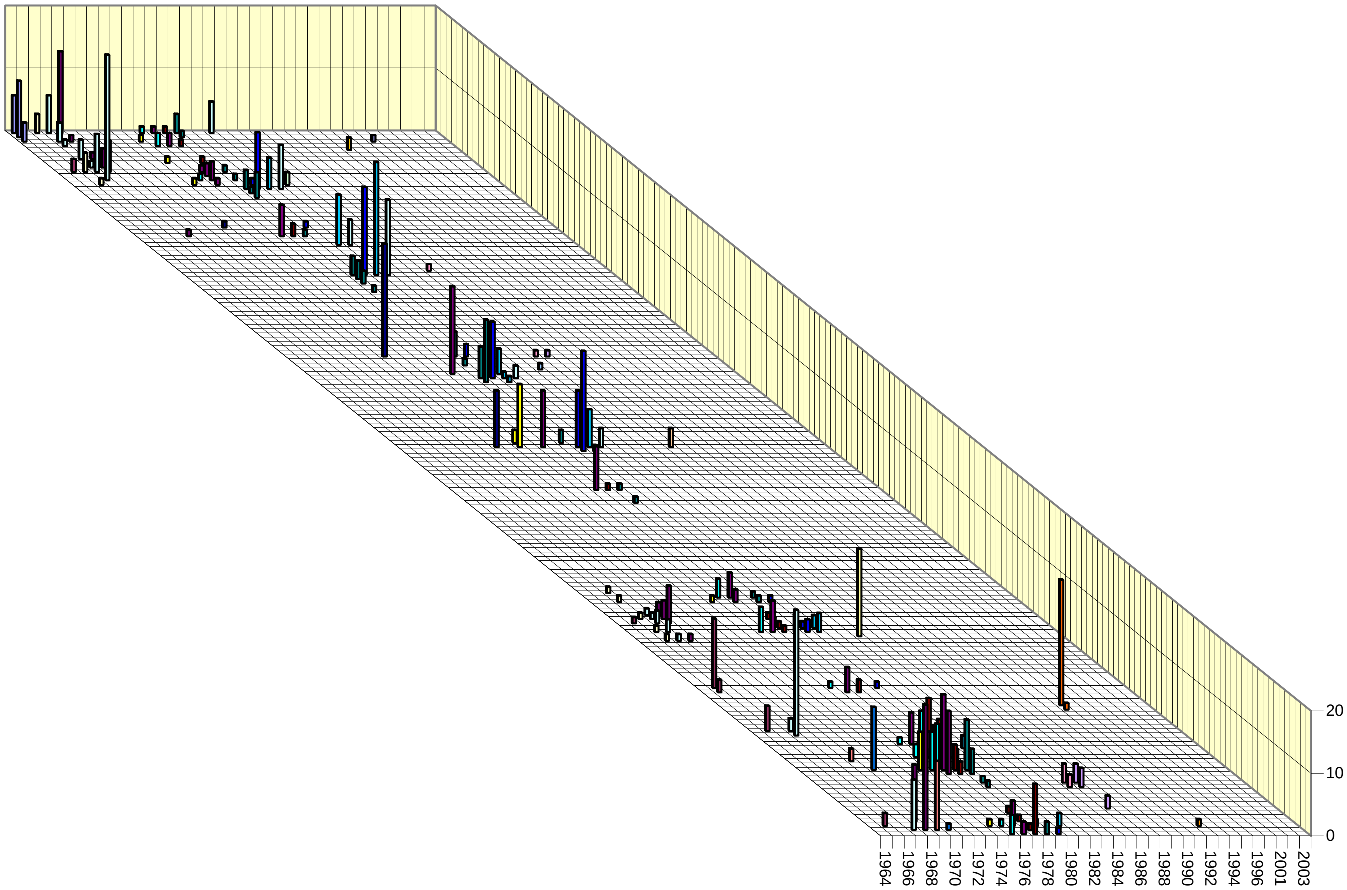



**Figure A.12: NT dataset – View of A.10 showing monthly counts.**