A REVIEW OF GUIDELINE DEVELOPMENT FOR SUSPENDED SOLIDS AND SALINITY IN TROPICAL RIVERS OF QUEENSLAND, AUSTRALIA

JE Dunlop[1,2,*], BJ Kefford[3], VH McNeil[1], GB McGregor[1], S Choy[1] and D Nugegoda[3]

[1] Department of Environment and Resource Management, 120 Meiers Road, Indooroopilly 4068, Queensland, Australia.

[2] Smart Water Research Facility, Griffith University, Gold Coast Campus, Qld 4222 Australia.

[3] Biotechnology and Environmental Biology, School of Applied Sciences, RMIT University, PO Box 71, Bundoora 3083, Victoria, Australia.

Manuscript received, 16/11/2008; accepted, 9/3/09.

ABSTRACT

Suspended solids and salinity are among the highest priority contaminants for the management of freshwater biodiversity and ecosystems in tropical Australia. Although they are natural constituents of aquatic ecosystems, when elevated above background concentrations they can have negative impacts. The Australian tropics experience intense seasonal rainfall patterns with distinct dry winters and wet summers, with tropical rivers often experiencing large episodic flow events in the wet season and low or no flows in the dry season. Both suspended solids and salinity represent a complex mixture of constituents that are influenced by a range of factors including catchment land use, geology, hydrology and geomorphology. Accurate and reliable water quality guidelines are an essential tool for their management. Given the highly variable nature of tropical rivers and of the stressors themselves, defining trigger values for them in tropical Australia presents a considerable challenge. This paper describes how salinity and sediment sources, mechanisms of transport and the characteristics of their constituents interrelate to influence their patterns of exposure and potential impact. The application of existing approaches to guideline development, including departure from reference condition and toxicological approaches are reviewed in light of the requirements for tropical rivers with recommendations given on the most appropriate techniques to develop guidelines at present. Although concentrations of suspended solids and salinity are highly variable within tropical rivers, their constituents tend to follow patterns that when modelled allow the identification of spatial grouping of relatively homogeneous sediment and salinity zones. These zones provide a basis for better defining regional reference ranges, which can inform the design of toxicity tests to determine the potential effects of suspended solids and salinity in a way that is representative of those zones.

Key words: salinity; sediment; turbidity; tropical rivers; water quality guidelines

INTRODUCTION

Increases in stream sedimentation and salinity are among the highest priority contaminants for the management of freshwater biodiversity and ecosystems in Australia (Lovett et al. 2007). Sediment run off has been rated as the third greatest priority for waterway management in Australia by the National River Contaminants Program (Land and Water Australia 2002) and is a high priority pollutant identified in the Reef Plan (2003). Salinity, suspended solids and deposited alluvial sediments are naturally-occurring and ubiquitous components of freshwater streams. However, it is well recognised in the scientific literature that elevated concentrations and durations of exposure can result in profound ecological impacts (e.g. Hart et al. 1991; Ryan 1991; Williams 2001; Kefford et al. 2002; James et al. 2003; Kefford et al. 2003a; Marshall and Bailey 2004; Waters 1995). Impacts from salinisation and sedimentation are a complex set of physical and ecological interactions that occur at a number of scales ranging from individual organisms to entire ecosystems.

In this paper, the area within Queensland that falls to the north of the Tropic of Capricorn is referred to as the 'tropics' (see Figure 1). Although there are some dry areas, the tropics mostly experience warm and humid conditions year round and do not experience the seasonal temperature extremes that temperate areas do. The Queensland tropics experience intense seasonal rainfall patterns with distinct dry winters and wet summers and can be broadly sub-divided into the 'wet tropics' and the 'dry tropics'. Wet (>1200 mm pa) and humid tropical climates predominate in the coastal band of far north of Queensland, with arid (< 250 mm pa) and semi arid (250 to 600 mm pa) regions prevalent in central and western Queensland including most areas west of the Great Dividing Range. In their natural state, rivers in tropical regions of Queensland are generally associated with low salinity and carry highly variable sediment loads. Tropical rivers of northern Australia have unique values and characteristics and are under increasing pressure due to environmental threats and human activities (van Dam et al. 2006). Accurate and reliable water quality guidelines are an essential tool for management of salinity and suspended solids. However, because of their inherent natural variability, developing guidelines for these stressors in tropical rivers presents significant challenges.

IMPACTS OF SALINITY AND SUSPENDED SOLIDS ON STREAM BIOTA

Salinity is an indicative measure of the total concentration of dissolved ions and is usually represented as measures of electrical conductivity (EC) or total dissolved salts (TDS) (see Williams and Sherwood 1994 for definitions). The major constituents of salinity in surface waters are sodium, calcium, magnesium, potassium, sulphate, carbonate,

*Author for correspondence, email: jason.dunlop@derm.qld.gov.au

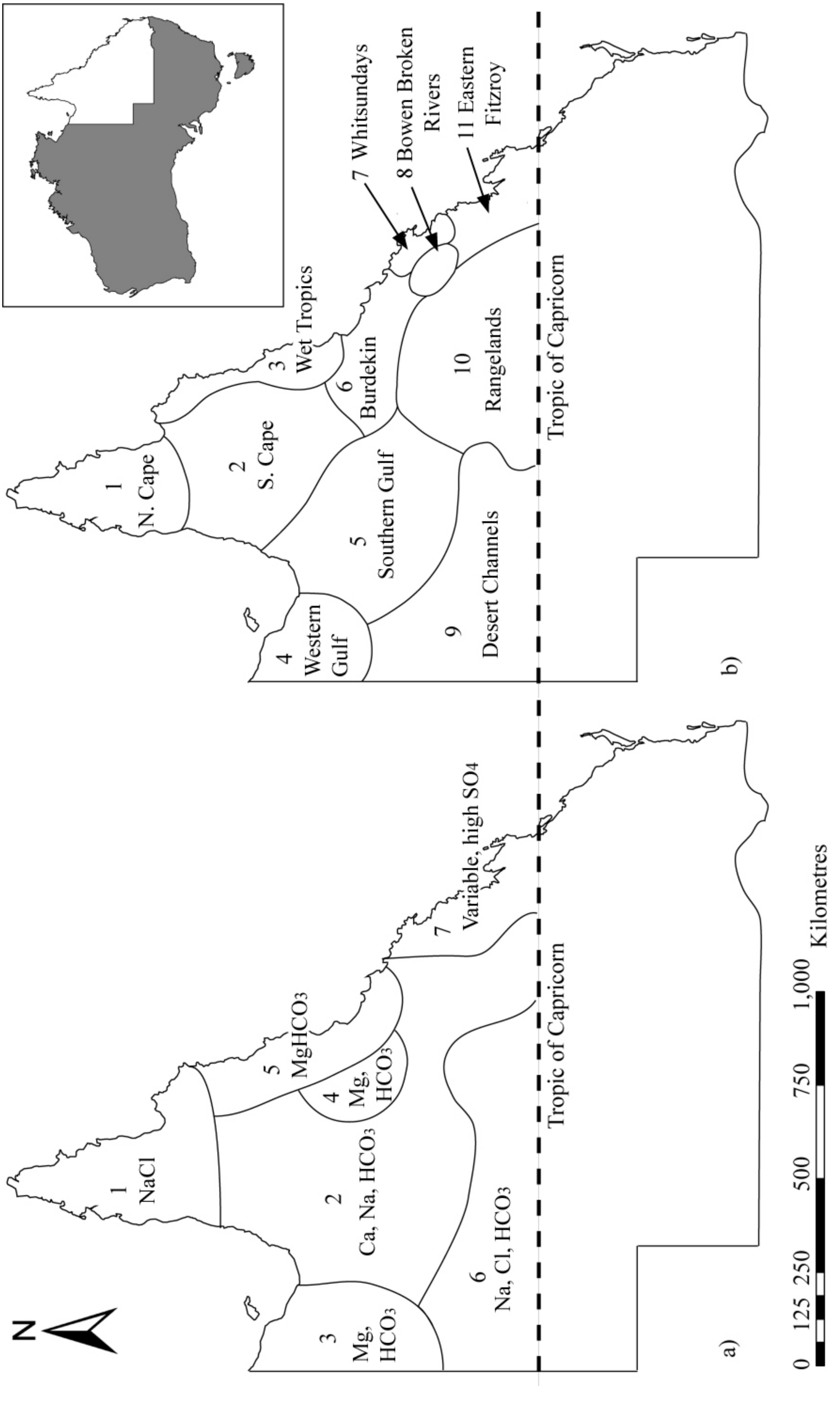

Figure 1. Stream salinity (a) and turbidity (b) provinces of Queensland (salinity provinces adapted from McNeil et al. 2005).

bicarbonate, and chloride. Soluble salts occur naturally in all aquatic ecosystems and are a vital component of the normal functioning of freshwater biota (Hart et al. 1991). Although salinity and sediments are natural components of freshwater ecosystems, they become contaminants when elevated above the normal ranges to which ecosystems have adapted. For all freshwater aquatic animals exposure to excessive concentrations of salt will have toxic effects, though a lack of salt can also act as a stressor for many species. Elevated salinity concentrations can cause mortality, or a reduction in growth (Boeuf and Payan 2001; Kefford et al. 2006a) and reproduction (Duncan 1996; Yang and He 1997; Kefford and Nugegoda 2005) or other sub-lethal responses (Hassell et al. 2006). Response may also occur at an ecosystem level resulting in changes in the structure of species composition (Metzeling 1993; Horrigan et al. 2005; Piscart et al. 2005). Structural changes can alter energy transfer pathways thereby affecting community functions (e.g. respiration and the breakdown of organic matter) and may cause shifts in populations (Boulton and Brock 1999; Nielson et al. 2003).

Tolerance to high salinity is in part due to the physiological mechanisms and morphological adaptations that act to balance internal concentrations of salts in the cells and tissue of an organism against the external environment. To maintain this balance, freshwater animals must regulate their bodily fluids to maintain an internal osmotic pressure greater than the outside environment. Tolerance is therefore dependent on the physiological and morphological characteristics that affect their ability to regulate ions and maintain homeostasis. The most salt-sensitive taxon tested in Queensland to date is Ephemeroptera Leptophlebiidae *Austrophlebioides* with an acute 72-hour LC50 of 6.9 mS cm^{-1} (Dunlop et al. 2008) though the mechanisms of salt toxicity are not well understood and it is not known why they are more salt sensitive than other groups.

Measures of total suspended solids (TSS) represent the dry weight per volume of particulate matter. TSS requires laboratory analysis to determine and are not able to be provided in real time. Turbidity can be measured in real time and is often used as a surrogate measure of TSS. Turbidity is the optical property of a liquid that causes light to be scattered and absorbed rather than transmitted in straight lines. It is thus an integrated measure of the suspended and dissolved load contributing to decreased water clarity and the properties of particulates held in solution (Bruton 1985). Substances that contribute to turbidity can include organic matter, phytoplankton, colour and mineral content. Turbidity is typically measured in Nephelometric Turbidity Units (NTU). Although turbid streams occur naturally in some parts of tropical Queensland (Natural Resources and Water 2005), when natural erosional processes are accelerated, streams experience increased turbidity and carry high sediment loads and concentrations (Dodds 2001).

Alluvial sediments occur as deposited and suspended solids. Although in some instances excessive TSS concentrations can cause a direct biological response, in most cases decreased water clarity from suspended solids has not been found to result in short-term acute mortality. However, there is considerable evidence for chronic, sub-lethal and indirect effects (see review by Dunlop et al. 2005 and the database of turbidity effects presented in the Appendix of Dunlop and McGregor 2007) though most of the available data has been collected for species found outside of tropical Australia. There are a large number of potential effects from stream sedimentation that include for example, responses relating to the smothering of benthic fauna or their habitat (Wood et al. 2005), reduction of photic depth and limited aquatic plant growth and stream productivity (Parkhill and Gulliver 2002), predator prey interactions (Abrahams and Kattenfeld, 1997; Granqvist and Matilla 2004), and increased drift of macroinvertebrates (Ryder 1989; Ryan 1991; Bond and Downes 2003). Although a decrease in stream productivity is listed as a potential effect of sedimentation, it is important to consider that some naturally turbid systems in western Queensland have been found to be highly productive under no flow conditions (Bunn and Davies, 1999; Bunn et al. 2003; Fellows et al. 2009).

SALINITY SOURCES AND VARIABILITY IN THE LANDSCAPE

Over time scales of hundreds of thousands of years and greater, salts have accumulated in the Australian landscape due to mineral weathering and the accumulation of dissolved salts from marine origins transported by rainwater. High evaporation and transpiration rates relative to surface and sub-surface runoff have prevented these accumulated salts from being effectively flushed, thus they accumulate in the landscape. This process has been termed primary salinisation. Secondary (i.e. anthropogenic) salinisation occurs when shallow water tables are artificially raised, dissolving the salt that has accumulated in the lower soil horizons carrying it towards the surface where it becomes concentrated through evapotranspiration. When irrigation does not contribute to the rising watertable the process is referred to as dryland salinisation. Other sources of salinity include marine water intrusion, irrigation salinisation, saline water discharges from mine pit dewatering, electricity generation, reverse osmosis water filtration, discharge of treated urban and industrial effluents and saline water disposal from salinity drainage programs in agricultural lands.

Surface water salinity and its ionic constituents are variable in tropical Queensland though they tend to follow patterns that indicate they are closely linked with the geology of, and water movement within, a catchment. McNeil et al. (2005) identified provinces with characteristic salinity ranges and ionic compositions within Queensland's flowing waters (Figure 1a). Streams in tropical northeast Queensland that drain short steep catchments with high rainfall generally have low salinity (median of <100 µS cm^{-1}, seldom exceeding 150 µS cm^{-1}) and are dominated by sodium chloride (McNeil et al. 2005). In the same study the proportion of sodium chloride was generally found to decrease westward from the east coast. Streams draining the low relief, semi arid, western side of the Great Dividing Range, or flowing into the southern Gulf of Carpentaria were found to contain relatively hard water (high Ca concentration) given the moderate salinity levels (ECs seldom exceeding 500 µS cm^{-1} with a median of approximately 250 µS cm^{-1}) with high proportions of magnesium in low salinity waters from areas within the vicinity of basalts (McNeil and Clarke 2004, McNeil et al. 2005).

The dominant ions in the salinity zones shown in Figure 1a are as follows; zone 1 is dominated by NaCl, zone 2 is dominated by Ca, Na, HCO_3, zones 3 and 4 are dominated by Mg and HCO_3, zone 5 is dominated by $MgHCO_3$, zone 6 is dominated by Na, Cl, and HCO_3, zone 7 has variable ions, with relatively high SO_4. Although the ionic composition of rivers in Queensland is variable, as the salinity of surface waters increases the ionic composition generally approximates that of sea water (McNeil et al. 2005).

Most instances of dryland salinity recorded in Queensland have occurred in the southern and central areas, usually at the edges of broad alluvial plains with low relief and moderate stream salinity (medians 200 to 400 µS cm^{-1}). These areas are usually associated with tree clearing. In tropical Australia, dry land salinity is not yet considered to be a major water quality issue for rivers, despite a relatively large amount of land in Queensland considered to be at risk of dry land salinity (ANRA 2003) and possibly climate change. However, localised instances of salinisation have been recorded in parts of tropical Queensland with occurrences recorded as far north as Weipa (Natural Resources and Water 2006). Long-term trends in salinity indicate that (between 1970 to 2000) mean stream EC across Queensland has not risen significantly and that the observed variability (range of mean ambient annual salinity of flowing waters in Queensland are 0.1 to 0.7 mS cm^{-1}) is mostly cyclical in nature and related to inter-decadal climatic fluctuations (McNeil and Cox 2007).

SEDIMENT SOURCES AND VARIABILITY IN THE LANDSCAPE

In tropical Queensland, the key anthropogenic sources of sediment to waterways are likely to be land clearing and degradation from over grazing and a loss of soil cover associated with broad scale cropping, though in some areas gully and stream bank erosion undoubtedly contribute large amount of sediments.

Sediments can originate from outside the river channel from colluvial processes or from alluvial processes within the channel itself. In addition to anthropogenic disturbance, sedimentation can be affected by a number of natural factors that include the erodability of the parent soil, rainfall and vegetation patterns, flow regimes, and geographical features (Wood and Armitage 1997).

In the same way that salinity follows patterns in the landscape so too does turbidity (Figure 1b). A modelling approach developed by McNeil (unpublished) identified zones of like turbidity in Queensland. As this approach has not been formally published elsewhere at this stage, it is summarised here in sufficient detail to describe the modeling process for the purpose of this discussion. This approach was developed using the Queensland Department of Natural Resources and Water's extensive SWAN (Surface Water Ambient Network) water quality database. Turbidity and flow values used in the models include all validated turbidity and flow recordings taken at gauging stations operated by the Queensland Department of Natural Resources and Water between 1965 and 2007. The names and locations of monitoring stations are detailed on the department's web site (Natural Resources and Water 2009).

To identify like turbidity zones it was necessary to partition the effect of two key factors known to influence turbidity: the catchment area upstream of the sampling location and the flow rate at which a sample is taken. To identify the turbidity zones the data were analysed in a series of steps. The first step was to classify all the turbidity values based on the stage of flow at which the turbidity sample was collected. These were regarded as flow categories of the turbidity values. To account for the effect that the catchment area upstream of the sampling location may have on turbidity, categories of turbidity values were subsequently established for catchment area in the same way as they were for the flow categories. Once categories for flow and catchment area were assigned to individual turbidity values it was possible to calculate percentiles from a distribution of turbidity values for each of the flow category classes within each catchment area category. These percentiles allowed each turbidity sample to be classified as low to high relative to the stage of flow at which the turbidity sample was collected and in relation to the catchment area upstream from the sampling point.

Finally, this information was then used to define the turbidity zones by clustering the proportion of samples regarded as high moderate or low turbidity in three separate flow categories (high flow, moderate flows and low flows). The spatial distribution of site clusters (or site groups) were examined with respect to the flows, catchment area, topography, geology and rainfall distribution that occurs within them. Provisional regions with similar turbidity characteristics were then manually refined where necessary to fit sites into the turbidity zones. The location of turbidity zones as identified by this process are shown in Figure 1b and are as follows; zone 1 is the Northern Cape, zone 2 is the Southern Cape, zone 3 is the Wet Tropics, zone 4 is the Western Gulf, zone 5 is the Southern Gulf, zone 6 is the Burdekin (upper), zone 7 is the Whitsundays, zone 8 is the Bowen Broken Rivers, zone 9 is the Desert Channels, zone 10 is the Rangelands, and zone 11 is the Eastern Fitzroy. The mean, lower 10^{th} and upper 90^{th} percentiles of turbidity values recorded for a range of flows in the eleven turbidity zones identified in Tropical Queensland including the number of sites and samples in each of the zones are displayed in Table 1.

Turbidity is a dynamic variable and can fluctuate naturally according to the stage of flow at which the sample has been recorded making it difficult to determine normal ranges without the consideration of flow. This is because a measure of turbidity is related to the amount of fluvial sediments held in the water column, which is highly dynamic and generally increases with discharge or until the supply of sediment becomes limited (Rossi et al. 2006). In general we can therefore expect higher turbidity levels during high flow conditions compared with base-flow conditions (Bilotta and Brazier 2008) though this relationship may not occur in streams where the supply of fine material is limited.

Table 1. Mean, lower 10th and upper 90th percentiles of turbidity values recorded for a range of flows in eleven turbidity zones in Tropical Queensland including the number of sites and samples in each.

Flow $m^3 s^{-1}$	Western Gulf	Southern Gulf	Northern Cape	Wet Tropics	Burdekin	Bowen Broken	Whitsunday	Eastern Fitzroy	Desert Channels	Rangelands	Southern Cape
0.002	5 (3,10)	20 (3,94)	16 (2,98)	4 (1,73)	6 (2,39)	6 (2,66)	3 (1,9)	7 (2,52)	127 (13,432)	53 (7,562)	8 (2,40)
0.01	11 (4,33)	12 (2,70)	7 (2,21)	3 (1,19)	5 (2,39)	5 (1,47)	3 (1,7)	4 (1,20)	132 (16,337)	30 (6,363)	5 (1,24)
0.03	10 (3,36)	10 (2,71)	4 (2,9)	3 (1,11)	5 (2,44)	4 (1,36)	3 (1,7)	3 (1,15)	133 (16,325)	27 (6,341)	4 (1,21)
0.08	8 (2,27)	11 (2,84)	3 (1,6)	3 (1,10)	6 (2,53)	4 (1,30)	2 (1,6)	3 (1,16)	132 (16,323)	29 (6,386)	4 (1,23)
0.2	6 (2,19)	13 (2,106)	2 (1,6)	3 (1,11)	7 (2,65)	4 (1,28)	2 (1,7)	4 (1,21)	131 (15,325)	36 (6,473)	5 (1,27)
0.5	5 (2,15)	16 (2,135)	2 (1,6)	3 (1,14)	10 (2,78)	4 (1,29)	2 (1,8)	5 (1,30)	131 (15,329)	46 (7,591)	6 (2,33)
1	5 (2,14)	20 (3,171)	2 (1,6)	3 (1,18)	12 (3,93)	4 (1,32)	2 (1,9)	6 (2,42)	131 (14,334)	59 (7,734)	7 (2,40)
2	5 (2,14)	24 (3,214)	2 (1,7)	3 (1,24)	16 (3,110)	5 (1,37)	3 (1,11)	8 (2,60)	132 (15,340)	75 (8,896)	8 (2,48)
3	6 (2,18)	30 (4,261)	3 (1,8)	4 (1,32)	20 (4,127)	6 (2,44)	3 (1,14)	10 (8,34)	133 (15,334)	94 (9,1073)	10 (3,58)
5	9 (3,25)	37 (4,316)	3 (1,10)	4 (1,41)	24 (5,147)	8 (2,56)	4 (1,17)	13 (3,117)	135 (16,344)	117 (9,1267)	12 (3,69)
9	14 (4,41)	46 (5,378)	4 (1,12)	5 (1,54)	30 (6,168)	11 (3,73)	5 (2,23)	17 (4,161)	138 (18,361)	144 (11,1478)	15 (4,81)
15	28 (6,84)	58 (7,451)	4 (1,14)	5 (1,70)	38 (7,193)	16 (3,103)	6 (2,32)	22 (6,225)	141 (20,389)	178 (12,1712)	19 (5,96)
25		75 (9,540)	5 (2,18)	6 (1,92)	49 (9,222)	28 (5,160)	10 (4,47)	31 (8,320)	146 (24,442)	222 (14,1969)	24 (7,113)
47		98 (12,646)	7 (2,23)	8 (2,122)	65 (12,259)	55 (8,284)	17 (6,79)	43 (10,466)	159 (31,548)	279 (17,2237)	32 (9,134)
95		135 (17,768)	10 (4,31)	11 (2,162)	89 (18,305)	144 (15,623)	38 (13,155)	63 (14,695)	216 (45,784)	351 (23,2475)	44 (13, 158)
215		191 (26,888)	17 (7,42)	17 (3,206	127 (28,362)		115 (38,390)	98 (20,1038)	358 (78,1374)	434 (32,2587)	62 (20,182)
560		279 (46,951)		32 (5,237)	185 (50,426)			157 (26,1484)	763 (177,4386)	509 (52,2416)	92 (33,197)
1700		408 (93,906)		77 (12,217)	272 (107,487)			252 (32,1854)		522 (101,1830)	137 (58,187)
Total sites	6	14	12	48	35	15	14	26	4	77	29
Total samples	146	424	319	3025	1824	673	580	1391	48	3595	1065

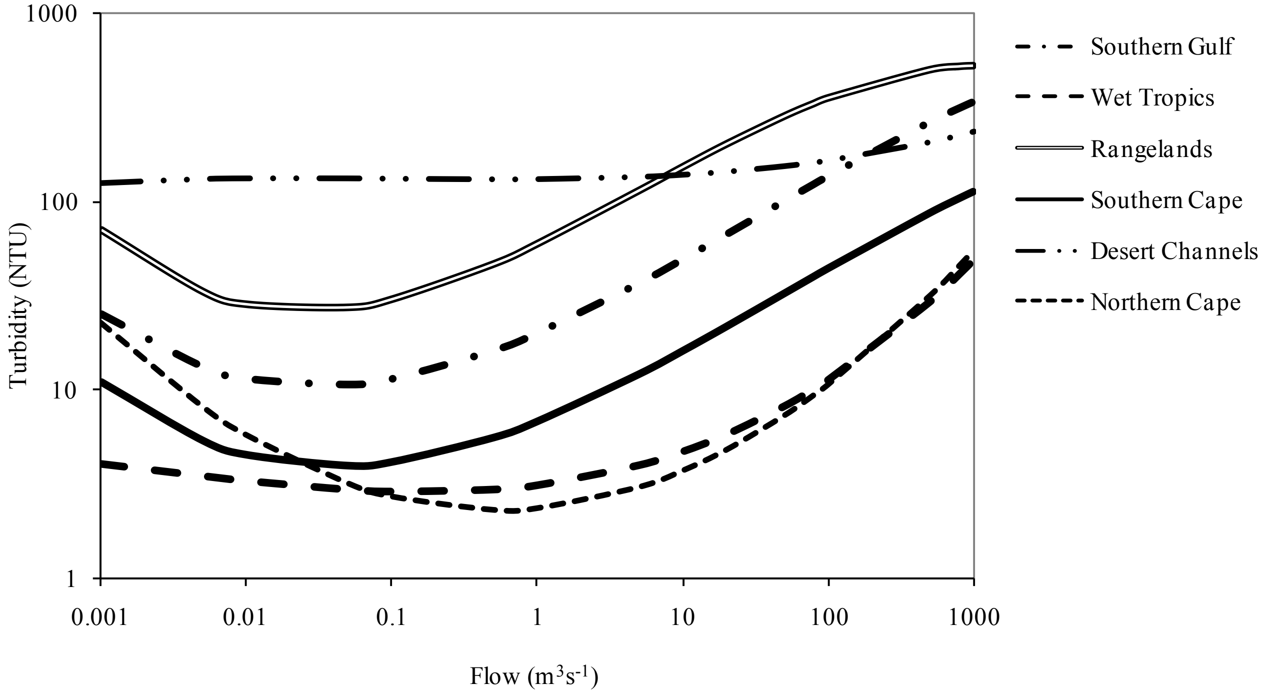

Figure 2. Relationship between turbidity and flow for six of the eleven turbidity zones in Tropical Queensland (The mean, lower 10th and upper 90th percentiles of the data used to derive these relationships are shown in Table 1).

Profiles of the turbidity/flow relationship in a selection of major river systems in Tropical Queensland are shown in Figure 2. These indicate that turbidity levels are highly variable between different river systems when there is no flow and indicate that turbidity has a zero or slightly negative correlation with flows up to approximately 1 $m^3 s^{-1}$ (except for the Desert Channels where fine sediment is likely to be more readily remobilised). Turbidity can be seen to increase above its no or low flow turbidity level with higher flows (Figure 2). We suggest that the negative association between flow and turbidity is due to the effect of initial dilution with slight increases in flow. Then with greater flows there is remobilisation of sediments once flow reaches a critical velocity, finally tending to a plateau determined by the supply of sediments. Turbidity in the Wet Tropics turbidity zone can be seen to be lower at low flows and has a relatively constant turbidity with increasing flow compared with other areas and only at high flows (approximately >10 $m^3 s^{-1}$) can be seen to increase. This is most likely due to differences in the relative availability of alluvial sediments that may be transported from the small, steep, high discharge catchments. This suggests that there is likely to be less deposited sediment that may readily be remobilised at lower flows and explains the relatively low background concentrations of TSS and turbidity compared with other areas of Queensland, and defines the conditions to which the stream and estuarine ecosystems have adapted.

Also, as these streams receive relatively regular baseflows, sediments within the smaller size fractions are likely to be flushed through the system more readily than the sediments with the same characteristics in the Condamine Uplands. In a study by Mitchell et al. (2007) of spatial and temporal trends of water quality in the Tully River, sediment supply was observed to be limited. The authors attributed this effect to the high density of ground cover vegetation that occurs in the Wet Tropics that effectively limits the amount of sediment able to be eroded before the supply is exhausted.

In general, tropical rivers flow intermittently, with most sediment transport occurring during large episodic flow events occurring during the wet season (Finlayson and McMahon 1988; Puckridge et al. 1998; Brodie and Mitchell 2006). Flow and discharge patterns can vary quite considerably between tropical rivers affecting their capacity to transport sediment making it useful to make comparisons between catchments based on annual sediment loads. For example, the Tully River has multiple major flows each year, compared with the Herbert and Pioneer Rivers which generally have one major annual flow, and with rivers in the dry tropics such as the Burdekin and the Fitzroy Rivers in which major flows are separated by periods of 4 to 10 years (Brodie and Mitchell 2006). A clear distinction divides the large 'dry' catchment rivers such as the Burdekin (with a catchment area of 133 000 km^2 and an annual mean discharge of 11 million ML) from the wet tropics rivers such as the Tully (with a catchment area of 2850 km^2 and an annual mean discharge 5.3 million ML) (Brodie and Mitchell 2006). Along the east coast of Tropical Queensland catchments with the largest areas produce the greatest sediment loads, with models indicating that the export of suspended solids to Great Barrier Reef Lagoon for the Burdekin and Fitzroy are estimated at 4000 kt a^{-1} (Hateley et al. 2006). However, when suspended solids generation is determined on a per unit area basis, the Mackay Whitsunday region was found to have the highest suspended solids generation figures (Hateley et al. 2006). In large flow events, the smaller Wet Tropics catchments can discharge most of their load of fine sediments and dissolved material completely to the estuary (Prosser 1996; Furnas and Mitchell 2001; Brodie and Mitchell 2006). Most of the sediment generated in Great Barrier Reef (GBR) catchments (as estimated by the long-term annual average SedNet model) occur within 120 km of the coast, typically where rainfall is highest and slopes steepest (Cogle et al. (eds) 2006).

CHALLENGES FOR DEVELOPING WATER QUALITY GUIDELINES FOR SALINITY AND SUSPENDED SOLIDS IN TROPICAL RIVERS

Although in temperate areas there is spatial and temporal variation in salinity, ionic composition and suspended solids, these factors are in general more pronounced in tropical regions. The need to incorporate this variability in the development of water quality guidelines for salinity and sediment in tropical regions poses some challenges. Included is the need to consider the natural changes in these constituents that occur due to seasonal flow regimes, catchment geology and stream geomorphology. Although either a departure from reference condition or a biological effect based trigger value (TV) may be used to define a trigger value, the use of an effect based TV is preferred by the national water quality guidelines (ANZECC / ARMCANZ 2000). There are several factors that make it difficult to develop high-reliability guidelines using toxicological approaches including:

- defining a suitable test medium applicable to a range of applications (particularly for sediments);
- a lack of effects data including sub-lethal chronic effects;
- accounting for the numerous site specific factors that can affect toxicity at the individual organism level through to ecosystem level impacts, particularly flow variability and low background concentrations; and
- in relation to sediments, the lack of a consistent concentration response relationship.

Although at present it is possible to derive a biological effect based guideline for salinity (e.g. Kefford et al. 2007), a lack of consistent effect data for suspended sediments makes it difficult to do the same for suspended sediments. An alternative means of deriving a guideline for sediments is a departure from reference condition approach. This provides a practical solution and although more widely used there are also a number of barriers that impede the development of reliable guidelines with this approach that include: a lack of suitable reference sites in many locations, the use of turbidity as a surrogate measure of sedimentation (alternatives are TSS or settling rates), and difficulty in defining a suitable reference range that limits the variability within the distribution of reference values. Where it is not possible to apply either of the aforementioned approaches, the national (ANZECC & ARMCANZ 2000) and Queensland water quality guidelines (QWQG) provide default triggers in the interim (QEPA 2006) though these cannot guarantee long-term protection due to the high level of natural variability within the regions.

Existing guidelines

Current water quality guidelines for salinity, turbidity and TSS are referential guidelines based on broad scale data sets. To improve their local relevance for tropical rivers there is a need to refine them according to localised factors affecting their variability in tropical Queensland. Factors requiring greater consideration when refining these guidelines include the expected variation caused by seasonal flows and high and low flows, and the expected longitudinal gradients of salinity and TSS that occur naturally between the upstream and downstream reaches of a stream. There is also a need to consider the effect of total loads on exposure patterns. Regional default trigger values recommended in the national guidelines do include a zone for Tropical Australia. The upper and lower ranges of the default regional trigger values for upland and lowland rivers as recommended in the national water quality guidelines for salinity and turbidity (for slightly to moderately disturbed ecosystems) recognise that guidelines for Tropical Australia require lower trigger values, less (20–250 µS/cm, and 2–15 NTU) than those specified for Southwest Australia (120–300 µS/cm and 10–20 NTU). The Queensland water quality guidelines (QWQG) provide greater resolution than the national guidelines for turbidity by considering upland and lowland categories separately. For example, TVs stipulated for the Wet Tropics are 6 and 15 NTU for upland and lowland areas respectively. Upland streams are expected to have turbidity values around half that expected to occur in lowland reaches of streams. Broad variation in the natural ranges of salinity are apparent in Queensland rivers, a phenomenon reflected in the recommendation of eighteen different salinity zones in the QWQG (QEPA 2006). The guidelines for salinity provided in QWQG (QEPA 2006) are not split into separate trigger values for upland and lowland rivers but rather upper and lower 75th and 25th percentiles of the eighteen zones. The default trigger values are provided as general, broad-scale guidelines that should be refined for local relevance.

As mentioned earlier, tropical Queensland experiences a distinct wet and dry season. The Douglas Shire Council Water Quality Improvement Plan found that ‘first flushes’ of sediment occurred at the start of the wet season and subsequent exhaustion of sediment supplies occur as the wet season progresses. As seasonal flow dynamics in the tropics have an appreciable effect on the duration, frequency of exposure and concentrations of sediments and also salinity (example provided in Hart et al. 1987), there is a need to develop separate wet and dry season guidelines. At present neither the national nor the Queensland default guidelines stipulate separate wet and dry season guidelines, although this is the case in the United States USEPA water quality criteria’s consideration of seasonal differences in flow conditions (an example can be seen in Alberta, Alberta Environment 1999).

The default trigger values are also not relevant for assessing TSS impacts during high flow events as they are relevant only for baseflow or ambient conditions and currently there are no guidelines that stipulate load based criteria. The Canadian Environmental Quality Guidelines provide a good example of how to address the issue of flow based criteria as they stipulate guidelines for high and low flow conditions based on flow class (CCME 2007). The Surface Water Assessment Network (SWAN) is a program that monitors ambient water quality in Queensland and is undertaken by the Queensland Department of Natural Resources and Water. As guidelines for total suspended solids are designed for application in ambient conditions, previous assessments of turbidity observations under the SWAN program have excluded extreme events from comparison with guidelines. However, as the bulk movement of sediment occurs during extreme events it may be important to consider TSS concentrations during events. This would require the development of guidelines for high flows, and although this is a notoriously difficult task, it may be important in representing natural variability. To improve the relevance of TSS guidelines under varied flow scenarios, Bilotta and Brazier (2008) suggest the use of a rating curve approach that relates TSS to discharge. A TSS/velocity relationship would seem a useful approach to account for the expected TSS concentrations under variable flows though this approach remains untested. In addition, the use of velocity would also provide some indication of the abrasive force of water in a flow event. The reason for this is that a given volume of water passing through a small channel at high velocity is likely to have a greater abrasive force and carry a higher sediment load than would the same flow through a large channel. Therefore, when defining reference ranges for TSS under different water flow scenarios, stream velocity would be a more appropriate variable for grouping sediment measures than would flow. However, given the difficulty associated with measuring water velocity it is often difficult to acquire accurate data across a large area and at the extremes of flow.

The existing guidelines may be improved by simply stipulating different wet and dry season guidelines for salinity, turbidity and TSS. In future it will also be beneficial to develop event based guidelines for turbidity and TSS that consider contaminant behaviour under flow conditions possibly using a relationship with water velocity though these relationships will differ for each river system and require further research. All of these suggestions are in line with the requirement for continual improvement in the resolution and the need to improve the local relevance of water quality guidelines.

Lack of a consistent dose-response relationship for sedimentation

Although in recent times considerable advances have been made evaluating the tolerance of freshwater taxa to salinity, there remains a lack of consistent tolerance information regarding sediment impacts and limited conceptual understanding that links it together. This makes it difficult to apply the existing effect data to the development of a water guideline for suspended and deposited sediments using a Species Sensitivity Distribution (SSD) approach that is applicable to a range of sedimentation scenarios. Some authors have questioned whether a traditional concentration response model is applicable at all for suspended solids and deposited sediments (Newcombe and MacDonald 1991; Bilotta and Brazier 2008). Newcombe and MacDonald (1991) suggested that the effects of suspended solids on aquatic biota cannot be predicted simply by using concentration data, and therefore concentration based guidelines in isolation from other sources of information may not be appropriate. Given the range of expected responses and the range of physical and chemical properties of suspended solids, it would be more appropriate to characterise them and the biota present at a location of interest and apply a conceptual response model to identify how the local taxa are likely to respond to the sedimentation occurring in that system.

Deposited and suspended sediment have many different types of impacts on freshwater biota. The conceptual model in Figure 3 shows some of the major mechanisms of sediment impacts in freshwater ecosystems and the linkages between impacts on habitat, individuals, populations and communities. Although there are a number of impacts associated with sedimentation, the primary impacts from which many other impacts arise are those associated with habitat disturbance including smothering of the benthos, reduction in water clarity and photic depth and increased erosivity and scouring of flows. When deriving biological effect based guidelines for sedimentation it will be important to ensure all three of these habitat impacts are represented in the response model that underpins any SSD used to derive a guideline. This would mean three separate guidelines would be developed for each of the primary impacts associated with sedimentation. Within this conceptual framework the effect of turbidity, TSS, sediment settling rates and bed disturbance frequency and scour may be monitored through the presence or absence of sensitive species that may be linked to expected effects. The conceptual model provides a basis to work with, though to improve the environmental relevance of any response model, particularly given the complexities of how sedimentation is likely to elicit impacts in aquatic ecosystems, further research effort is required to develop and test this conceptual model and investigate biological responses to suspended sediments both in the laboratory and in the field.

An approach to investigate potential impacts on populations is to examine the change in abundances (see Leung et al. 2005; Kwok et al. 2008) or occurrence of species as sediment increases to develop field based SSDs. However, such an approach makes the assumption that changes in species abundances or occurrences are in fact caused by the changes in salinity or sediment and not by (unknown) confounding factors. A novel approach to assess the potential hazard of sediments by determining a proportion of species at risk is described in reports by Parametrix (1999), Smith (2000) and Smith and Markham (2000). In this approach the likely sensitivity of fish species to specific modes of action is scored from 0 to 3 for specified TSS ranges. The natural distribution of fishes is then used to derive sensitivity curves based on inherent sensitivities. A species with a rank of 3 for any factor or a rank sum of 5 or more is considered potentially at risk for that TSS range. This approach has been successfully applied and related to observations of fish populations along a gradient of sedimentation (Smith and Markham 2000).

Defining a suitable test medium

In the case of salinity, a standard composition of salt (often reconstituted or synthetic marine salts) provides a uniform test medium to simulate salinity impacts, and a significant amount of effect data has been generated in this way (Kefford et al. 2003b; Allan 2006; Dunlop et al. 2008). Although (as

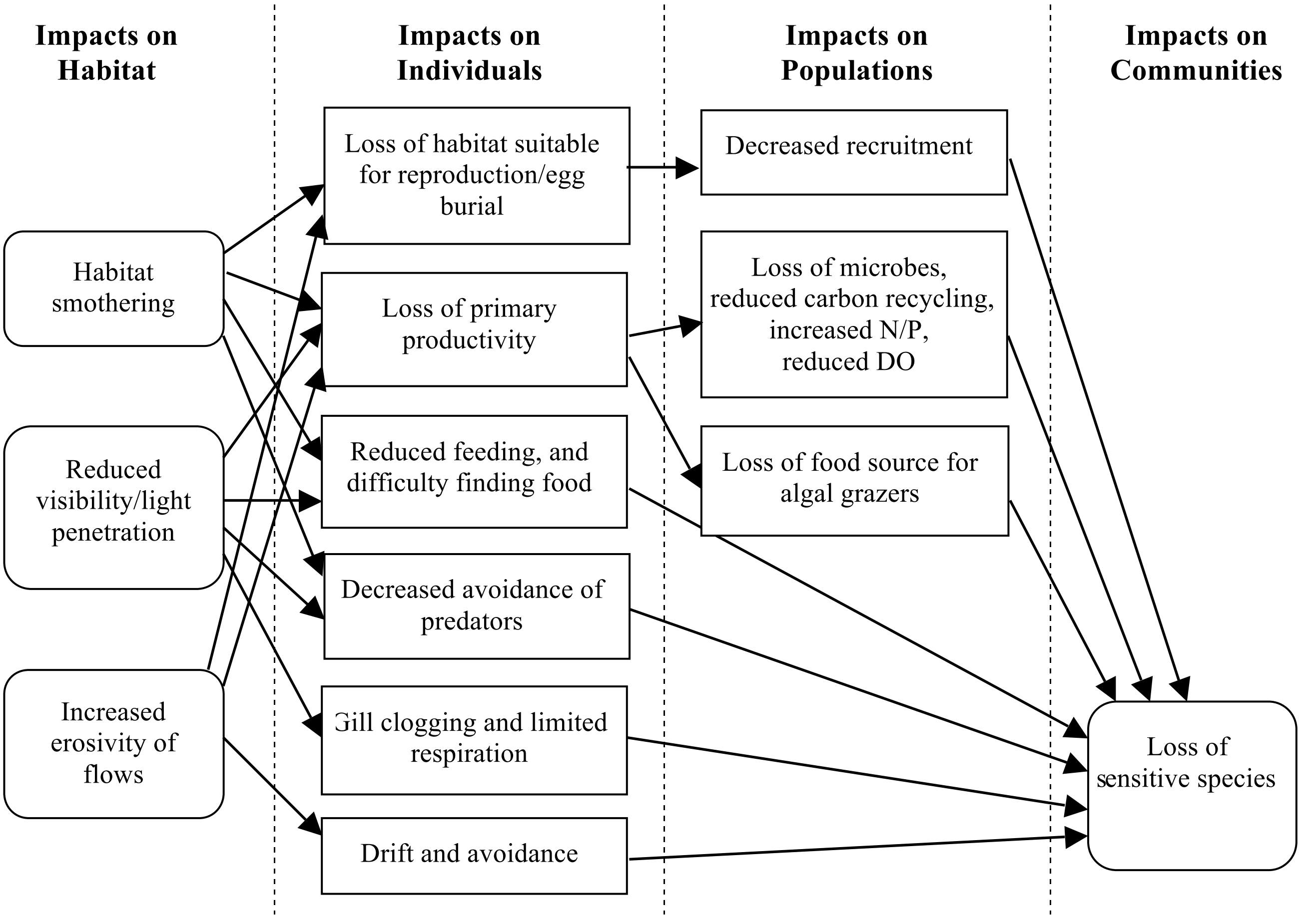

Figure 3. Conceptual model of the mechanisms of impacts from sedimentation of freshwater aquatic ecosystems.

previously discussed) ionic composition is known to vary in the landscape as salinity increases, the ionic composition of natural waters tends to approach that of sea water. This makes concentration-response tests using marine salts generally applicable to predicting the effects of salinity increases in surface waters. However, variations in ionic compositions are known to result in differences in biological responses compared with that of seawater (Mount et al. 1997; Zalizniak et al. 2006; Dunlop and McGregor 2007) and where ionic composition differs from that of seawater the effect of variable ionic compositions should be evaluated. Marine salts may also not be ideal to assess the effects of point source discharges of saline waters from, for example, mines and electricity generation, as these can have variable ionic compositions and associated characteristics such as pH (see Zalizniak et al. 2009) which are markedly different from that of natural surface waters. Care should be taken to ensure that the effects of the specific ionic composition are carefully considered in the development of water quality guidelines in these scenarios.

It is notoriously difficult to define a standard test medium for sediments that is applicable to the wide range of sediment impact scenarios that occur in the environment. This is primarily due to the highly variable properties of suspended particulates that contribute to measures of turbidity. Without a standard definition of what constitutes suspended solids or turbidity, it is not possible to undertake comparative toxicity studies that are applicable to a wide range of environments. In addition, suspended solids can have indirect effects on populations by altering habitat through abrasion, smothering and the loss of primary productivity. Sediment exposure can occur over different durations and have variable return frequencies although the effect of sedimentation occurring across multiple flow events are not well understood. Therefore traditional concentration-response experiments are not likely to be indicative of all the effects of sediments and tolerance ranges for TSS and turbidity determined in laboratory experiments provide only a limited explanation of the potential impacts. Although sediment is highly spatially and temporally variable, it is possible to use standard sediments for toxicity testing and many different test media have been used to assess sediment impacts. Some examples of the many test media used to represent sediment impacts in effect experiments include kaolin, illite, sericite, clays, silts, sand, river sediment, wood fibre, diatomaceous earth, and bentonite clay (see review by Dunlop and McGregor 2007 for examples cited in the literature). Whilst the use of these standardised test media for turbidity provides a basis for comparison between species effects for a given test medium, the temporal and spatial variability of sediments mean that the effects observed are not likely to have wide applicability to compositions other than that of the standard tested. In addition to a lack of consistency in the media used to undertake tests, there is also a large array of measurement of effects used including a range of non-typical end-points. Sediment effect data are often not reported as estimates of NOEC/LOEC or LC/EC50 and there are only point estimates for a few species reported in the literature making it difficult to populate a SSD for sediments.

Lack of sub-lethal effects data

Although considerable progress has been made in the last decade on understanding how salinity and sediment impact freshwater biodiversity, there is very little sensitivity sub-lethal or chronic data for either of the stressors and this lack of data is particularly apparent for tropical Australian rivers. Much of the experimental work done to date on salinity impacts have been short-term acute tests conducted in temperate regions, although comparisons of 72-h LC50 values between macroinvertebrates collected from Victoria (Kefford et al. 2003b, 2006b), Tasmania (Allen 2006) and Queensland suggest that this work is broadly applicable to sub-tropical and tropical rivers (Dunlop et al. 2008). Without sub-lethal and chronic data it is necessary to apply either a generic acute to chronic ratio, or a safety factor to account for sub-lethal effects. Using data from south-east Australia, Kefford et al. (2006b and 2007) advocate the use of safety factors calculated from the loss of species richness with increasing salinity as predicted by a SSD and the actual loss of species richness across pooled samples (Kefford et al. 2006b, unpublished data, respectively). Whilst such an approach is useful it does not replace the requirement for laboratory-derived sub-lethal and chronic data.

Accounting for locally relevant factors

In addition to the difficulty associated with defining an appropriate test medium, test duration and exposure conditions, predicting the effects of TSS and salinity is likely to be moderated by the same environmental and chemical factors that influence the impacts of all contaminants in aquatic ecosystems (including temperature, pH, hardness, and the presence of other contaminants such as nutrients and pesticides) that may alter impacts. Sediment can also transport a large volume of organic contaminants such as nutrients and pesticides adsorbed to particle surfaces. In some cases the combined effects of salinity, sediments and other associated contaminants needs to be considered. Suspended solids are known to precipitate out of solution when combined with high salinity (Grace et al. 1997; Donnelly et al. 1997) and as this can increase the rate of sedimentation, the combined effect of suspended solids in the presence of salinity is likely to compound their individual effects. As salinity and sediments are naturally-occurring components of aquatic ecosystems, there is difficulty in separating elevated concentrations from background conditions. An effect of prior exposure to salinity has been observed to increase tolerance (see for example reviews by Hart et al. 1991; James et al. 2003).

Defining a suitable reference range

For a referential approach to provide sufficient protection, it requires enough data from a reference site or sites to account for the natural variability of the area in question. The default guidelines provided in the QWQG (QEPA 2006) provide reference ranges for turbidity that are defined according to geographic units corresponding to major drainage divisions. In some instances, areas of variable geomorphology and hydrology occur within them therefore areas of like turbidity do not necessarily fit major drainage divisions. Their subsequent use in defining a reference range can increase the resultant variability of the reference range distribution. To limit this variability and improve the reliability of guidelines there is a need for regional reference ranges to reflect the sources of variability in the landscape. For example, turbidity and sedimentation naturally increase along a longitudinal gradient from a rivers source to its estuary. To account for this McNeil et al. (2000) recommends developing background limits for different reaches within a basin as well as regional/catchment guidelines. Preliminary work by McNeil et al. (2000), Thomson and Marshall (2004) and Dunlop and McGregor (2007) support the method of using a modeling approach to identify groups of sites with similar water quality, rather than determining guidelines on a regional basis and according to upland/lowland position in catchment.

This type of approach is similar in concept to that applied by the European Union (EU) Water Framework Directive that defines protective concentrations for waters based on their typology. Typology is defined according to latitude, longitude, altitude, depth, geology and size. It is also related to the natural hydro-morphological and physico-chemical conditions including the power of the water body to erode sediments and the size and availability of sediments (resistant rocks compared with more easily displaced sands and silts) (UKTAG 2007, 2008). Although these have not yet been adopted, this framework provides a logical and meaningful way of managing each system based on its inherent characteristics. Application of this approach to tropical Queensland would require modification as it tends to limit the availability of monitoring data that may be used to define a reference range within each of the defined spatial areas.

DISCUSSION

There are no simple solutions to deriving water guidelines for salinity or suspended solids. With the current information available we recommend the use of a biological effect based approach for salinity coupled with a referential approach to check and where necessary, improve the local relevance and reliability of salinity guidelines. Full application of a biological effects approach requires much more information to be collected on the sub-lethal and chronic impacts of salinity, especially in tropical Australia where high reliability guidelines often are required. Also, further information is needed to investigate the differences in toxicity associated with variable compositions of major ions. This is most likely to be of concern for point source discharges of saline water as their compositions are likely to vary from those of natural waters.

Due to the difficulty in defining a test medium applicable to a range of exposure scenarios and because the results of sediment tests do not typically follow standard dose-response relationships, at this stage there is not sufficient dose response information to derive a biological effect based approach for sediment. There is also a lack of effect data that are able to adequately represent the range of impacts associated with sedimentation and in particular there is a lack of effect data for tropical species. There is subsequently a need for improved conceptual understanding of the impact of sediments in aquatic ecosystems and toxicological research to test their validity and populate effect models. In lieu of the effect data required to derive a guideline representative of all potential ecological responses sedimentation, we propose that guidelines be developed for suspended solids applying a variation on the 'departure from reference' approach to where the reference ranges are defined according to the natural variability in the landscape rather than catchment boundaries alone. A reference range approach is the only practical method of setting trigger values for TSS and turbidity at present, so it is important that reference ranges are defined in such a way that limits the variability of the reference range distribution used to produce interim triggers with a reasonable degree of reliability. While this approach provides a sound solution in the short term, it does not directly consider the effect of habitat smothering and erosivity of sediment-laden flows. Future improvement to sedimentation guidelines should ensure all potential responses are represented in the response models. Whilst the application of a reference based approach provides an interim solution for turbidity and TSS, its application requires the collection of more reference data with greater spatial and temporal resolution than is currently available, particularly in the Western Cape and Gulf provinces where monitoring data are sparse. However, it is difficult to find reference sites in unimpacted areas as much of the landscape has experienced changes from that of natural conditions and rapid land use change has occurred in many previously undeveloped areas. For this reason long-term solutions to identifying TSS and turbidity guidelines will most likely be found through the development of ecotoxicological effect data for sediments.

One problem that remains with the collection of effect data for salinity and sedimentation is that as toxicological effects data are derived under standard conditions, in many cases the effect data may not match the variations in sediment and salinity constituents and durations and frequency of exposures that occur naturally in the environment. Although it is not possible to undertake toxicity testing for all possible exposure scenarios, it may be possible to design future toxicological experiments around the conditions likely to be experienced within unique spatial areas of like sedimentation or salinity characteristics such as the salinity and turbidity zones presented here. Defining what makes one zone different from another may help to determine exposure regimes and test media that will allow extrapolation of results to those unique zones. Application of the approaches recommended here would therefore lead to the development of a combined referential/biological effects methodology to derive guidelines.

ACKNOWLEDGMENTS

This paper was written as part of an Australian Research Council (ARC) Linkage project (LP0669113). Kefford was supported by an Australian Postdoctoral Industry (APDI) fellowship from this grant. We thank a number of reviewers for their comments on the manuscript including Nina Polaschek, Barry Hart and an anonymous reviewer. We also thank Rick van Dam for his encouragement throughout the writing of this paper.

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