internal report 346

Effects of suspended solids on macroinvertebrate communities downstream of the Jabiluka Mine

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November 2000

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EFFECTS OF SUSPENDED SOLIDS ON MACROINVERTEBRATE COMMUNITIES DOWNSTREAM OF THE JABILUKA MINE

A thesis submitted by Gael Davidson, B. Env. Sc., in partial fulfillment
of the requirements for the Degree of Bachelor of Science with Honors
in the School of Biological, Environmental and Chemical Sciences,
Faculty of Science, Information Technology and Education

Northern Territory University
November 2000

Certificate of Authorship of Thesis

I declare that this thesis is my own work and has not been submitted in any form for another degree or diploma at any university or other institute of teriary education. Information derived from the published and unpublished work of others has been acknowledged in the text and a list of references given.

Signature of Author ....

Date ....

ACKNOWLEDGEMENTS

This project has been completed with the generous assistance of a number of people and I would particularly like to thank them:

My supervisors, Chris Humphrey (***eriss***) and Michael Douglas (NTU) for their encouragement, guidance and patient assistance through every phase of the project. In particular, Chris Humphrey for giving me the opportunity to see Kakadu National Park, an unforgettable place. I am also grateful to the Center for Tropical Wetlands Management (NTU) for providing me with a scholarship.

James Boyden assisted with every facet of the project, from data analysis to field work, to document preparation. David Norton provided methodical instruction on bug identification, good-humored company in the lab, and many hours performing the water chemistry analysis, along with Karen Rusten and Dene Moliere. Matt Webb assisted early on with the field work, and sensibly, entered the GPS points for 7J Creek. Tony Mount provided ready assistance with computer queries, and Ben Bayliss cooked the best stir-frys in Jabiru.

Kieth McGuiness from NTU provided understanding of, and patient assistance with, the statistical analysis of the data. Michael Welch from NTU helped with sampling sorting and identification, and did not miss any nematodes. Peter Dostine and Cyrus Edwards helped out with bug identification, and, Robert Batterham from EWLS assisted with the field work with unending enthusiasm, and provided many insightful suggestions on data analysis.

And finally, my family, for all their support.

ABSTRACT

Construction of the Jabiluka mine in July 1998 created the potential for elevated levels of suspended sediment to wash into Swift Creek via several tributaries, over the 1998/99 wet season. To assess disturbance on the benthic macroinvertebrate fauna in Swift Creek, downstream of the mine, a modified MBACIP design was implemented. Paired sites were selected in Swift Creek (upstream of mine influence and downstream with potential mine influence), and in three control streams: Catfish, North Magela, and 7J creeks with placement of upstream/downstream sites all approximately 1 km apart. An additional downstream site was also sampled in Swift Creek to detect any gradient effects if present. Sampling was undertaken every 3 weeks over the wet season, commencing in late December until early May, when recessional flows commenced.

Turbidity recorded at the gauging stations in Swift Creek indicated elevated turbidity at the downstream site relative to the upstream site for the first 6 weeks of sampling. However, maximum levels recorded were relatively low and short-lived in contrast to earlier studies where effects were observed. The modified MBACIP design allowed for comparisons to be made between streams based on multivariate dissimilarity, and univariate difference (richness and abundance) data.

Results found no anomalies in Swift Creek in relation to the control streams for any of the measures of site difference. ANOVA between creeks for dissimilarity and difference data revealed no significant interactions between the creeks at any of the sampling events (apart from those attributable to higher macroinvertebrate abundance at Catfish Creek). Macroinvertebrate community structure over the wet season produced arch-shaped trajectories for all creeks, indicating similar seasonal patterns of taxa succession.

Therefore, results all support a conclusion of no observable effects on the macroinvertebrate communities downstream of the Jabiluka project area.

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Chapter 1: Introduction

This study examines disturbance in streams through the responses of benthic macroinvertebrate fauna over the 1998/99 wet season, in the Northern Territory, Australia. Trends of recolonisation and faunal succession are compared between four streams to test for anomalies in one potentially-impacted stream, which flows downstream from the Jabiluka mine site. The following review provides information on the role of disturbance in aquatic ecosystems, and reviews information relevant to the ecology of seasonally-flowing streams and impact assessment in lotic systems.

1.1 Disturbance in streams

Stream communities experience a range of natural environmental extremes that can be physical, chemical and/or biological (Doeg et al 1989). Physical extremes include stream discharge where high flow events can dislodge large stones causing saltation, scouring of the stream bed, and an increase in sediment loads to the water column and/or benthos. Stream flow influences many other variables such as habitat area, current velocity, channel geomorphology and substratum stability (Poff and Ward 1989). High flow events can clearly disturb the environment and, when they affect stream fauna detrimentally can be thought of as a "disturbance", defined as "any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability or the physical environment" (Pickett and White 1985). Communities have often been found to respond in a predictable way to environmental disturbances, indicating that they may be a necessary factor in maintaining community diversity (Pickett and White 1985).

Resh et al (1988) discuss the way that macroinvertebrate communities are adapted to their local, seasonal and environmental range and therefore qualified the definition of a disturbance by suggesting that it must be an event that is unpredictable and not typical of the seasonal range. A recent study from Nepal found that highly seasonal intense monsoon rains did not appear to constitute a disturbance, as benthic abundance was already low prior to the rains and recolonisation was rapid (Brewin et al 2000). Death and Winterbourne (1994) and Poff (1992) both expressed concern at the difficulty in assessing just what qualifies as an unpredictable or atypical event, concluding that "physical measures" are required, regardless of the predictability of the event. In other words, there must be some measurable cause (e.g. sediment de-stabilisation), or effect (e.g. noticeable loss of taxa abundance) for an event to qualify as a disturbance.

Townsend et al (1997) suggest that an appropriate way to avoid the confusion around the notion of predictability is to define and measure disturbance in an "organism-related sense". The *response* from the macroinvertebrate community can therefore reflect the severity of a disturbance (Yount and Niemi 1990). Patterns of recovery by the faunal community present a way to measure or quantify a disturbance by possibly correlating any trends to some measurable physical effect (Townsend et al 1997) such as discharge (Poff and Ward 1989), or amount of substrate movement (Death and Winterbourne 1994; Cobb et al 1992).

1.2 Effects of suspended sediments on aquatic ecosystems

Many human activities such as logging, construction, urbanisation and mining, disturb the environment, creating a situation where soil and sediment are de-stabilised and readily washed into aquatic ecosystems such as rivers and streams. Sediment addition to aquatic waterways has been described as one of the most serious and widespread anthropogenic impacts (Hynes 1970).

1.2.1 Relationship between suspended sediment and turbidity

Turbidity values were used to estimate suspended solid loads in Swift Creek and the Control streams. Turbidity and suspended solids are related, even though they actually quantify different properties of a water sample. Turbidity (NTU) measures scattering and absorption of light, a direct result of the concentration, size and shape of the suspended particles, together with their refractive index (G Dunkerley pers. comm). Tannins and other humic substances in the water column can also alter the optical properties of water. Suspended solids (mg/L) measures the mass of the insoluble particles remaining on a filter paper of a given pore size.

Gippel (1988) mentions that turbidity can be a good surrogate measure of suspended solids concentration owing to a well defined functional relationship that exists between light scattering and particle concentration, and suggests that in an ideal suspension, we would find a linear relationship between the two. However, seasonal and periodic storm events will produce differences in sediment properties, such as their reflective or absorptive properties, and in background water colour and turbulence. These confounding effects can alter the suspended sediment-turbidity relationship (Gippel 1989).

A strong relationship between suspended sediment and turbidity will occur, therefore, when the particle size distribution remains relatively constant and several studies have found this to be the case. However, these relationships are very site specific (eg the Namoi River, NSW; several lake systems in the central highlands of Tasmania) (G Dunkerley pers. comm). A reasonably good relationship between turbidity and suspended solids was found by Stowar (1997) in Jim Jim Creek, in Kakadu National Park

The following Table 1.1 has been adapted from Stowar (1997) showing a summary of the findings from selected studies on suspended sediment and their effects. A full review can be found as Appendix 1.

1.2.2 Summary of main findings of previous studies

These previous studies indicate that, to produce a response in the macroinvertebrate community, levels of suspended sediment must be both reasonably prolonged (as either a pulse or sustained effect over several months or several years), and must reach higher than normal maximum levels for most of this time. In general, time frames extended well over two months. Even when sediment addition was compounded 20-fold over a two month duration, recovery of the taxa affected took only 3 weeks (Gray and Ward 1982). Clearly, macroinvertebrates can be highly resilient and recover quickly when the disturbance ceases, providing that both intensity and duration are not too severe, and the recolonisation source is not adversely affected.

Table 1.1 Summary of observations reported by selected studies on the effects of suspended sediment on stream macroinvertebrate communities

Location and Reference	**Nature of disturbance**	**Exposure**		**Observed effects on benthic macroinvertebrates**
		Max Turb./S.sol. (background)	Duration	
Jim Jim Creek, Kakadu NP, NT Stowar (1997)	Elevated turbidity downstream of a road crossing	60 NTU/100 mg/L (<5 NTU/10mg/L	3 months	Macroinvertebrate community changes; probably reduced taxa abundane, primarily Chironomidae
SE NSW, Australia Richardson (1985)	Elevated turbidity and sedimentation resulting from forestry activities	1800 NTU, 30.1 NTU ave (1.3 NTU)	8 months	Reduced abundance of selected taxa, increased invertebrate drift
Victoria, Australia Chessman et al (1987)	Elevated suspended sediment plus sedimentation associated with dam construction	480 NTU (generally low, but range 1-110 NTU)	Pulses over 3 years	Reduced abundance of a range of species
South West WA, Australia. Growns & Davis (1994)	Suspended inorganic solids associated with forestry.	60 mg/L (<10 mg/L)	4 months	Mean species richness decreased, mean total taxa abundance decreased, though not statistically significant.
ACT, Australia. Hogg & Norris (1991).	Elevated suspended solids following storms, resulting from urban development.	560 mg/L (generally < 5 mg/L).	Brief pulses over several years prior to study	Reduced species richness and macroinvertebrate density
New Zealand Quinn et al (1992)	Clay discharges from mining activities	154 NTU (< 8.2 NTU)	Mining for 2-8 years prior to study	Reduced invertebrate densities downstream (by 9-45%).
North Carolina, U.S.A. Lemly (1982)	Elevated suspended sediment and sedimentation associated with runoff from logged and residential development areas.	2360 mg/L (generally <5 mg/L).	Not specified	Reduced species richness, abundance and biomass of filter feeding taxa.
Wyoming, U.S.A. Gray & Ward (1982)	20-fold increase in suspended sediment, no appreciable sediment deposition	> 300 mg/L (<20 mg/L).	2 months	Densities of some taxa decreased (including chironomids); some increased (eg oligochaetes, others unchanged.
Colorado, U.S.A. Cline et al (1982).	Elevation of suspended solids associated with road construction activities.	Range of max levels among sites: 70 – 500 mg/L (<10 mg/L).	4 months	Reduced density, abundance and diversity of macroinvertebrate community.
Ontario, Canada. Barton (1977).	Short term elevation of suspended solids from highway construction	1390 mg/L (<5 mg/L).	8 months	Altered species composition, no change in total abundance or species richness.

1.2.3 Adverse effects from suspended sediment on aquatic biota

Macroinvertebrates, fish and plants can be affected by suspended sediment through a variety of mechanisms, either direct (i.e. physiological) or indirect (e.g. alteration of habitat). A summary of these potential effects is presented in Table 1.2. and a full review is found as Appendix 2.

Table 1.2 Summary of adverse effects upon aquatic biota from suspended sediment

Major Taxanomic group	**Specific effect**
Macroinvertebrate grazers	Settled sediment can cover periphyton thereby reducing food supply (Ryan 1991, Newcombe and McDonald 1991) causing a decrease in abundance. Respiratory structures may be effected in many species (Metzeling et al 1995).
Macroinvertebrate filter feeders	Gills can become clogged and/or food supply dominated by inert matter causing starvation (Hellawell 1986; Newcombe and McDonald 1991) resulting in a decrease in abundance.
Hyporrheic taxa eg Oligochaetes and some Chironomidae species	Interstitial spaces can become clogged thereby reducing available habitat (Richards and Bacon 1994) causing the taxa to burrow further into this zone or drift, with a resulting decrease in abundance. In contrast, an increase in abundance has been noted in Oligochaetes (Gray and Ward 1982).
Ovipositers eg Simuliidae larvae	Substrate can become scoured or unstable making attachment difficult (Chutter 1968; Hellawell 1986) also resulting in drift and a decrease in abundance.
Fish	Suspended particles reduce feeding and respiratory efficiency causing stress (Hellawell 1986). Sighting prey can be more difficult (Reynolds et al 1989). Interstitial pores for egg laying can become covered (Peters 1967). Avoidance and altered feeding patterns may result.
Periphyton and Macrophytes	Light reduction can cause a substantial loss of primary productivity; 3-13% at 5 NTU (Lloyd 1987) and scouring of the stream bed can reduce plant biomass.

In summary it can be seen that macroinvertebrates are ideal indicators for detecting and assessing the potential disturbance of sediment addition. These organisms occupy a diverse range of habitats within the benthos, thereby being able to elicit a variety of responses, and they are clearly highly sensitive to sediment addition. Reduced species richness, abundance and/or changes to community structure are all potential consequences of high sediment loads in streams. These responses are generally rapid so that they can be detected over a short period of time. Recovery generally requires recruitment from upstream or nearby perennial sources (Paltridge et al 1997), creating a time lag before the community returns to pre-disturbance levels. This time lag is much reduced with mobile organisms such as fish, thereby potentially confounding interpretations of recent disturbance.

Fish may also exhibit avoidance behaviour, or stress-related responses that only appear in the longer term (Hellawell 1986); both may pass undetected within a 3-weekly sampling regime. Aquatic plants and algae also show responses to increased levels of suspended sediment (Lloyd et al 1987), but because of their sedentary nature, cannot record responses to increased levels of suspended sediment as rapidly as macroinvertebrates which may drift downstream in response to disturbance.

Benthic macroinvertebrates have several other important attributes making them ideal for biomonitoring programs:

- They are ubiquitous and found generally in high abundances in streams and rivers (Williams 1980; Hellawell 1986); statistical analysis of the data is, therefore, easily facilitated (Hellawell 1986).
- The different taxa vary widely in their sensitivities to human disturbance and will respond to a range of impact types (Rosenberg and Resh 1993).
- Macroinvertebrates are relatively sedentary and have life cycles ranging from several weeks to several years (Marchant 1986). This means that both temporal and spatial information on impact and community recovery can be obtained (Hellawell 1986; Chessman 1995;).
- Field collecting techniques are simple and easily undertaken, and identification with taxonomic keys is now quite well developed and user friendly.

Benthic macroinvertebrates have been used extensively and successfully in the ARR as bio-indicators for several years (Faith et al 1991; Faith et al 1995; Humphrey et al 1995; O'Connor et al 1995; Smith et al 1993; Stowar 1997). However, the main disadvantage of using macroinvertebrates for biological monitoring relates to the labour intensive nature of sample processing - sorting and identification. A further disadvantage of adopting benthic invertebrates for biological monitoring is that community structure can be influenced by factors other than water quality, including high discharge events such as floods, or highly variable seasonal changes (Rosenberg and Resh 1993). However, this problem is not unique to macroinvertebrates and also confounds programs using other groups of aquatic organisms. Therefore, proper inference about impact relies on well-designed studies that account for spatial and temporal variation.

1.3 Biomonitoring as an impact detection tool

There are two main approaches to biomonitoring of aquatic macroinvertebrates that can be taken, as outlined by Humphrey et al (1999). These depend upon the requirements of the monitoring program:

- Early detection of change requires rapid responses from organisms sensitive to the impact.
- Assessment of the ecological importance of any observed change requires measurement of important species or communities of organisms, able to provide a "picture" of ecosystem-level change.

This study focuses on the second approach, which is longer term monitoring to assess changes in community structure over time. These changes can then be analysed to infer impact arising from specific environmental disturbances. To this end, a monitoring program must be based upon a statistical design that will enable a suitable hypothesis to be tested.

1.3.1 An idealised monitoring approach

Biological monitoring programs are designed to detect impacts on the environment, and therefore knowledge about the natural environment prior to the disturbance is considered extremely important (Green 1979). This led to the development of the Before-After-Control-Impact class of designs: BACI (Green 1979). This basic monitoring design can be modified in several ways to increase statistical power; either temporally, as suggested by Stewart-Oaten et al (1986) with the BACIP design ("P" indicates paired sampling), or both temporally and spatially as recommended by Keogh and Mapstone (1995) with their MBACI design ("M" indicates multiple control locations). Whether or not a disturbance alters temporal and spatial variability of populations, or mean abundance, the inclusion of more than one control location will always better characterise the natural variability of the region, and thereby lead to stronger inferences about impact (Underwood 1991, 1993, 1994; Chapman et al 1995). Underwood (1994) also highlights that simultaneous paired sampling (P) is no longer a strict requirement, making for easier logistical planning.

Concerns associated with MBACI designs center on the use of several control locations. The geographical areas from where the control locations are selected may introduce such natural variability to the data that an impact passes undetected (Humphrey et al 1995). Faith et al (1995), as a way of compensating for this potential problem, suggested sampling from paired locations, upstream and downstream, within each of the control (non-impact) and impact streams (see below). This design is hereafter termed MBACI-P, referring to multiple (M) controls and paired (P) sampling from upstream and downstream sites in the same streams.

The BACIP or MBACI-P designs rely upon "*d*" or difference values calculated between the control and impact locations over time. Univariate data, based on total abundance and richness, or abundance of selected taxon, can provide this comparison (Stewart-Oaten et al 1986, 1992). However, benthic communities, like most faunal communities, comprise complex arrays of interacting species and therefore community data are inherently multivariate. Analysing community data sets as a whole, means that all the factors which create the community complexity contribute to the statistical picture or signal (Faith et al 1991; Clarke 1993; Jackson, 1993; Clarke and Warwick 1994).

Data from the upstream and downstream sites at each stream in an MBACI-P design, can be used to generate dissimilarity (from multivariate analysis) or *difference* values (from univariate analysis) which can then be compared between *control* versus *impact* streams for *before* versus *after* interactions, using asymmetrical ANOVA (Humphrey et al 1995). An impact would be reflected by a greater change in dissimilarity values for the impact stream (or *difference* values from univariate measures), over the control streams for the post-impact period (ANZECC and ARMCANZ 2000). The potential advantage of MBACI-P over MBACI designs is the greater constancy of the "difference" values at control streams, and at the impact stream prior to impact. The lotic influences acting upon each stream become incorporated into the design, reducing the natural variability of the region into stream "behaviour". Thus, there is potentially greater statistical sensitivity in MBACI-P designs and they are considered the ideal approach to impact detection (Humphrey et al 1995; ANZECC and ARMCANZ 2000).

One of the assumptions that must be met in any statistical analysis is that of temporal independence of the data (Underwood 1997). When this assumption is violated, generally when sampling occurs too frequently, influencing the dissimilarity or *d* values, then trends can become apparent in the data (Stewart-Oaten et al 1992; Keogh and Mapstone 1995). One option is to analyse the trend as part of a time series, as outlined by Chatfield (1984), and introduce the causative factor as a covariate. For example, Stowar (1997) found strong unidirectional influence on BACIP dissimilarity values as creek flow receded after the wet season in ARR streams (i.e. dissimilarity values declined with decreasing creek flow). This variable, (creek flow) was then treated as a covariate in a regression analysis and used to infer impact.

1.3.2 Where pre-impact data are not available

The idealised approach to impact detection described above could not be employed for the present Jabiluka study because of the lack of *before* or pre-impact data. A number of alternative approaches to drawing inference, within the MBACI-P design, may nevertheless be adopted, as described in the following sections.

1.3.2.1 Strict hypothesis testing procedures

While statistical inference is reduced considerably where pre-impact data are unavailable, Thrush et al (1994) outlined the usefulness of time series data when pre-impact assessment is not possible. The time series generated provides a range of values displaying the behaviour of the non-impact and impact creeks over the study period. Data from the control and impact streams can be analysed within an asymmetrical ANOVA to test for any statistically significant results, *post-impact*. ANOVA can utilise both multivariate and univariate (total taxa abundance and richness) data. Using multiple controls reduces the likelihood that all controls would undergo simultaneous fluctuations in the same direction. Therefore, there is greater statistical certainty about attributing changes in dissimilarity or *d* values to a disturbance (Keough and Mapstone, 1995).

However, it must be borne in mind that in any statistical analysis of impact versus control site data where pre-impact data are absent, the untestable assumption is made that the indicator, in this case, macroinvertebrate community structure, responded similarly in control and impact areas prior to the impact.

1.3.2.2. Multivariate ordination

Multivariate ordination provides powerful corroborative evidence to classic hypothesis-testing procedures used in impact detection studies. An ordination pattern represents the sampling sites in Euclidean space where the actual dissimilarity between sites, generated through their biological attributes (multivariate taxa abundance), is reflected in their distance apart in the ordination (Faith and Norris 1989).

Multivariate ordinations, importantly, produce temporal biological descriptions of sampling sites over time, and in this manner, trends between potentially impacted and control streams can be evaluated.

1.3.2.3 Correlates of ordination space

Environmental (including physico-chemical) variables can be incorporated into the ordination through Principal Axis Coordination (PCC), whereby vectors are produced showing direction of influence upon the ordination (Belbin 1993). This analysis can also be undertaken for all taxa within the biological community. A descriptive ordination is therefore produced, highlighting the most influential variables and taxa, and thereby providing a very powerful corroborative technique for an MBACI-P design. *Before* versus *after* ordinations can show if the biological spacing of an impact location changed relative to the other sites, and the variables, if any, which correlate to that change. When there is no pre-impact data, *after* only ordinations can show seasonal trends for different sites over time.

1.3.2.4 Disturbance gradient analysis

A gradient effect of disturbance can be assessed through the inclusion of additional sites in the treatment stream. If it can be shown that effects dissipate at increasing distance downstream of a point-source disturbance, this provides evidence that the disturbance was the cause of the impact.

1.4 Detecting an impact within a variable seasonal environment

Impact detection relies on distinguishing changes in the ecology of a system that are due to anthropogenic effects from changes due to natural seasonal patterns. In order to best infer a conclusion of either impact, or, no-impact, it is important to have an understanding of the environmental fluctuations and trends that characterise a particular environment, in this case, the aquatic ecology of lotic, intermittent streams.

1.4.1 Seasonal trends in macroinvertebrate community structure in intermittent streams

There is a notable paucity of studies on seasonal changes to macroinvertebrate communities in tropical, intermittent streams. In seasonally-flowing streams in Kakadu, N.T., there have been three studies which have examined the seasonal changes in macroinvertebrate communities over a wet season. Two of these were short term studies over one season or part thereof; one describing initial recolonisation trends after re-wetting (Paltridge et al 1997), and the second analysing ongoing seasonal patterns (Tripodi 1996). Douglas (1999), over two consecutive wet seasons, examined macroinvertebrate communities at two spatial scales (streams, and pools within streams) in two small, intermittent streams in Kakadu.

Most studies of seasonal changes in macroinvertebrate communities in intermittent streams have been undertaken in temperate locations such as the Werribee and Lerderderg Rivers in southern central Victoria (Boulton and Lake 1992a; 1992b; 1992c) or in desert streams such Sycamore Creek, Arizona (Gray and Fisher 1981; Boulton et al 1992a). Ordination patterns from the studies spanning several years (Douglas 1999;

Boulton and Lake 1992b,c) show that similar seasonal trends occur in the macroinvertebrate communities as they recolonise the streams after the dry period.

Boulton and Lake (1992b) describe for multivariate ordinations of the macroinvertebrate communities, that the stream which flowed continuously produced a "doughnut shaped" trajectory. This pattern represents a seasonal progression back to a similar biological community. The stream which underwent a dry period produced an arch-shaped pattern, not completing the full circle, indicating a break in flow conditions. This "arch" pattern was also found in the intermittent Kapalga streams studies by Douglas (1999).

Boulton et al (1992a) examined macroinvertebrate community structure in a Sonoran desert stream over two years and found that the fauna exhibited consistent seasonal patterns, despite a variable flow regime. Douglas (1999) also found consistent trends between seasons for colonisation at both streams. These studies would appear to indicate predictability in the recolonisation patterns. The consistent recolonisation and successional trends of the fauna are responsible for producing the intermittent (arch) or continual (doughnut) trajectories found by Douglas (1999) and Boulton and Lake (1992b) in ordination space.

Different taxa appear to benefit from the variable flow conditions that are found in intermittent streams. Boulton and Lake (1992c) described three main phases of taxa colonisation as "pioneer, mid-successional and summer taxa" and their dominance corresponds to the three main flow periods of early re-wetting, mid-flow and diminishing flow respectively. In this and other studies, the pioneer taxa are dominated by groups that are able to survive well over the dry season (Douglas 1999; Paltridge et al 1997). The variable flow conditions which ensue after initial re-wetting encourage a variety of very adaptable taxa (Boulton & Lake 1992c) and, ultimately, diminishing flow conditions create small pools in backwater areas, where prey can become concentrated, prompting an increase in the more predatory taxa (Douglas 1999; Boulton and Lake 1992c; Tripodi 1996).

In summary, there are clear trends in the seasonal patterns of macroinvertebrate community structure over a wet season from intermittent stream studies. While variability between locations, and between sites within the same studies, is evident

(Douglas 1999; Tripodi 1996), these earlier studies provide a sound background against which to assess the trends found in this study.

1.5 Background on Jabiluka

Kakadu National Park, located in the wet-dry tropics of Northern Australia, is Australia's largest National Park, covering an extensive 19,804 km^2 (see Figure 2.1). The Park's western boundary is located 120 km east of Darwin placing the entire area within the ecologically-diverse Alligator Rivers Region (ARR) where four major rivers drain northwards into Van Diemen Gulf. One of these rivers, the South Alligator, runs from headwaters to floodplain entirely within the boundaries of Kakadu, representing an entire catchment.

The diversity of natural ecosystems found within the Park includes mudflats, mangrove swamps, floodplains and billabongs, patches of remnant rainforest and open savanna woodlands. The terrestrial and aquatic ecosystems of the ancient sandstone plateau and escarpment region of the Park are particularly diverse and contain many endemic and relictual elements. Kakadu is exceptional within Australia in still possessing nearly all the plant and animal species that are believed to have been present in the area 200 years ago, even though, now, some of these are under threat (ANCA 1999). Seasonally-inundated wetlands comprise over one-fifth of the Park and are of major importance to migratory waterfowl and are protected under the International Ramsar Convention (Wellings 1995). The continual Aboriginal occupation of the Park for at least 50,000 years has enabled Kakadu to receive World Heritage listing for both cultural and natural heritage values.

The Alligator Rivers Region has rich mineral reserves and uranium is one of several important minerals located in the catchments. Both the Jabiluka and Ranger uranium ore bodies were discovered in the late 1960's. These abutt one another and lie close to the north-eastern boundary of the Park within the Magela Creek catchment. Stage One of Kakadu National Park was declared in 1979. However, it was not until 1984 that the Second stage expanded the Park to encompass the Jabiluka and Ranger mineral leases (Press and Lawrence 1995). Construction of the Jabiluka mine (Fig 2.3) commenced in

July 1998 resulting in 31.5 hectares of land being cleared and altered within the Swift Creek catchment which drains into the Magela floodplain (Kinhill and ERAES 1996).

Disturbance of the soil during the construction phase at Jabiluka created the potential for sediment to be washed downstream into the main tributary of Swift Creek with the onset of the first wet season flows in December, 1998, and for the remainder of the wet season. Increased suspended sediment and turbidity in streams, with resulting loss of water clarity, can have adverse effects upon aquatic biota (see Table 1.2). Compliance with stringent water quality standards is of the utmost importance in such an undisturbed and highly valued environment. One of the Commonwealth Environment Ministerial recommendations for the Jabiluka mine states that turbidity values in Swift Creek must not exceed one standard deviation of a baseline mean (Environment Australia 1996). However, increasing importance is now being placed upon biological criteria as indicators of ecological integrity in ecosystems, rather than changes in physico-chemical characteristics alone (Norris and Norris 1995; ANZECC and ARMCANZ 2000). Responses of biological indicators, such as aquatic fauna, provide key management goals for the assessment of environmental disturbance, reflecting the desired state of an ecosystem (Karr 1999). Therefore, if biological indicators, such as macroinvertebrate communities, were to reveal an impact, this would override the physico-chemical criteria (ANZECC and ARMCANZ 2000).

The present study was undertaken in the 1998-99 wet season (December 1998 to May 1999) and examined the seasonal and spatial dynamics of macroinvertebrate communities in several streams adjacent to the developing Jabiluka mine. The data were used as part of baseline information required to detect and assess potential mining impact. In particular, the ecological information was gathered using an experimental design that would best enable an assessment to be made of whether or not the current mine-related disturbance was adversely affecting the biota in adjacent Swift Creek that receives runoff from the Jabiluka project area.

1.6 General Aims of the study

The general aims of the study are as follows:

1. To test the null hypothesis I that "there is no difference in suspended sediment loads between Swift Creek and the control streams over the wet season".

2. To test the null hypothesis II that "there is no effect upon the aquatic macroinvertebrate communities in Swift Creek downstream of the Jabiluka mine site as a result of mine construction activities".

3. (i) To examine the seasonal patterns of fauna succession in the four streams, one "potentially-disturbed" stream (Swift Creek) and three control streams, to serve as a baseline from which to assess for any potential disturbance;

   (ii) Use such patterns and relationships to interpret and explain any changes observed in macroinvertebrate communities of Swift Creek downstream of the Jabiluka mine site in relation to the other sites;

   (iii) To assess the importance of any observed changes in the structure of macroinvertebrate communities at the Swift Creek downstream sites in relation to the other sites.

4. To assess the inferential capacity of a MBACIP design with few if any baseline or pre-disturbance data, in detecting potential impacts in streams arising from increased suspended solid loads.

5. To make recommendations for future study, and ongoing monitoring at Jabiluka.

Chapter 2: Materials and Methods

2.1 Study Area

2.1.1 Climate in General

This study was undertaken in the Alligator Rivers Region (ARR) of the Northern Territory, within the boundaries of Kakadu National Park (Fig 2.1). The climate of Kakadu National Park is monsoonal, with two distinct seasons: wet and dry. The wet, humid season extends from November to March and the cooler dry season from May to September, with slight variations occurring from year to year (Russell-Smith et al 1995). April and October are the transitional months. The climate is characterised by high annual mean temperatures ranging from 22.4 - 34.0° C with high humidity during the wet season. Mean annual rainfall ranges from about 1300 mm in the south of the Park to 1565 mm in Jabiru township (Russell-Smith et al 1995). The contrasting climatic changes between wet and dry seasons produce a varied and highly seasonal but predictable hydrological regime in the ARR resulting in high diversity within the aquatic ecosystems, both temporally and spatially (Corbett 1996).

The seasonal rainfall pattern described above limits the period of significant flow of most streams within the ARR to between the months of January to May. The discharge pattern over the wet season in the lowland stream channels is one of a series of flood spates resulting from heavy rainfall events, super-imposed upon a steady base flow. The larger streams, such as the South, East and West Alligator Rivers, flow year round in their lower reaches, (tidal reaches). However, in the smaller tributaries, such as Magela Creek, flow ceases during the dry season, apart from the upper, spring-fed sections of streams close to the escarpment (Humphrey et al 1990).

2.1.2 Description of study area

Swift Creek (potentially-impacted creek) together with 2 of the control streams, 7J and North Magela, are located within the Magela Creek catchment. These creeks form part of an eastern drainage system with their source in the (Arnhemland) plateau or dissected sandstone country and escarpment. Swift Creek receives additional runoff from the large

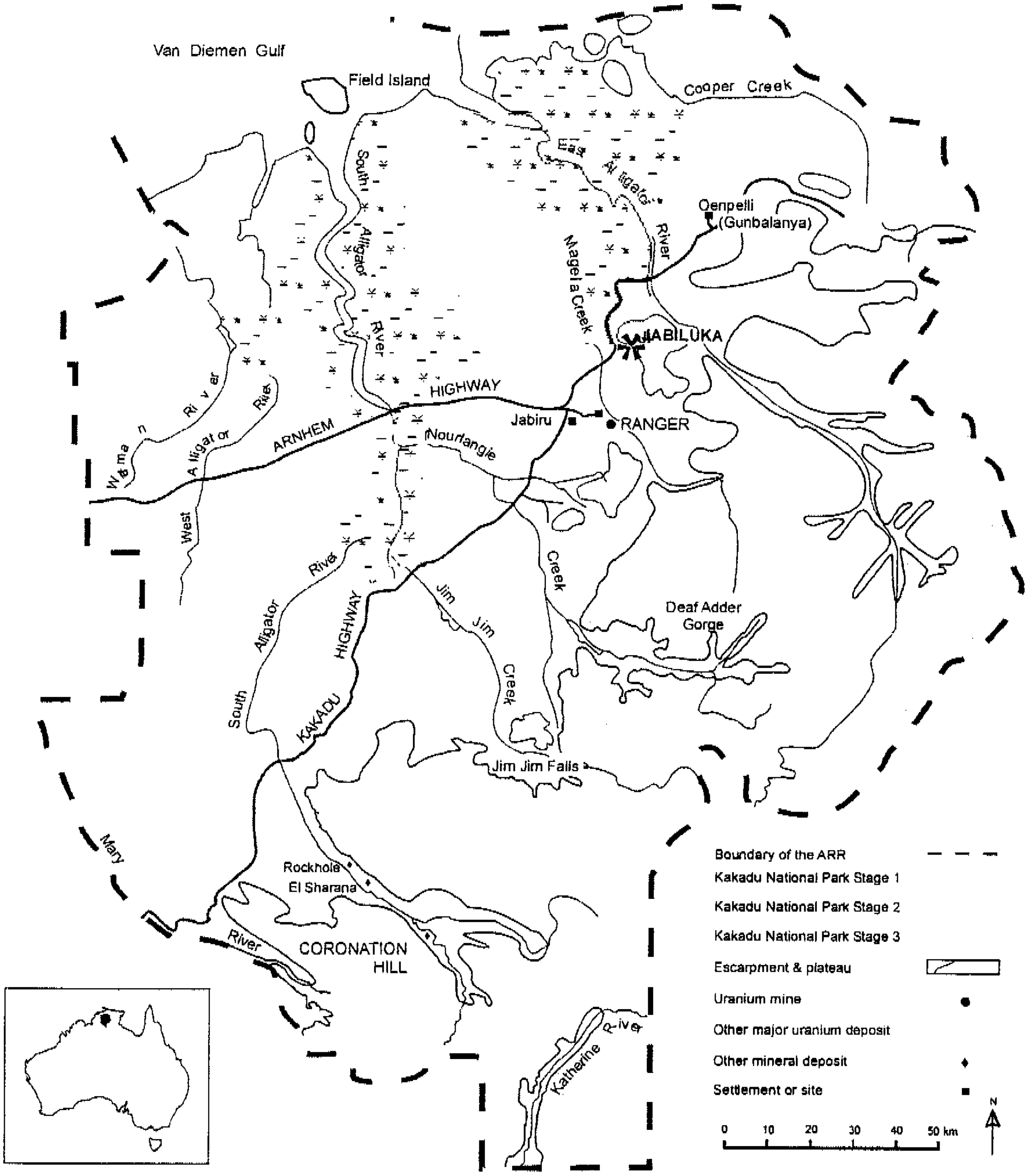


Figure 2.1 Location of the proposed Jabiluka mine within Kakadu National Park, Northern Territory, Australia

sandstone Jabiluka outlier, located on the Jabiluka mine lease (Fig 2.2). The three creeks emerge from deeply incised narrow gorges, to enter the lowlands as shallow, anastomising, alluvial channels containing sandy deposits. These lowland channels are prone to flooding during the wet season and vast amounts of sand on the creek beds are shifted each year. Eventually, the streams drain into the extensive floodplain or associated lagoons of the Magela Creek catchment. The creeks flow only during the wet season, commencing in about December, with recessional flows beginning sometime in April. Small pools can remain in some of the deeper channels throughout the dry season but water quality declines severely and the pools can become anoxic.

Catfish Creek, the third control stream, is located in a separate catchment to the north-east, draining the same dissected sandstone country as the three streams described above (Fig 2.2). It is a small tributary of the East Alligator River that is spring-fed from the base of the escarpment. For a short distance below the escarpment it probably flows year round (C Humphrey, ***eriss***, pers. comm). At the beginning of the sampling period in late December 1998, Catfish Creek had well-established macrophyte cover along the stream edges, indicating that it had been flowing for some time, in contrast to the other creeks.

At the lowland sites where sampling was conducted, the four creeks flow through largely open savanna woodland and forest dominated by *Eucalyptus* spp over a sparse layer of shrubs and tall grasses. These grasses, dominated by *Sorghum* spp and *Heteropogon triticeus* attained heights of up to 3 m by the late wet season. Riparian vegetation comprised *Melaleuca* spp, *Lophopetalum arnhemicum*, *Grevillea* spp and *Pandanus spiralis* interspersed occasionally with *Syzygium* spp and *Carallia brachiata*, and the freshwater mangroves, *Barringtonia acutangula*. Additional rainforest species were present in Swift Creek, including *Calophyllum sil*, *Maranthes corymbosa*, *Litsea glutinosa* and, *Xanthostemon eucalyptoides* and *Allosyncarpia ternata* at the upstream main channel.

Aquatic macrophytes established soon after wet season flows commenced in December. The main species present throughout the wet season were *Limnophila* spp, *Blyxa* sp,

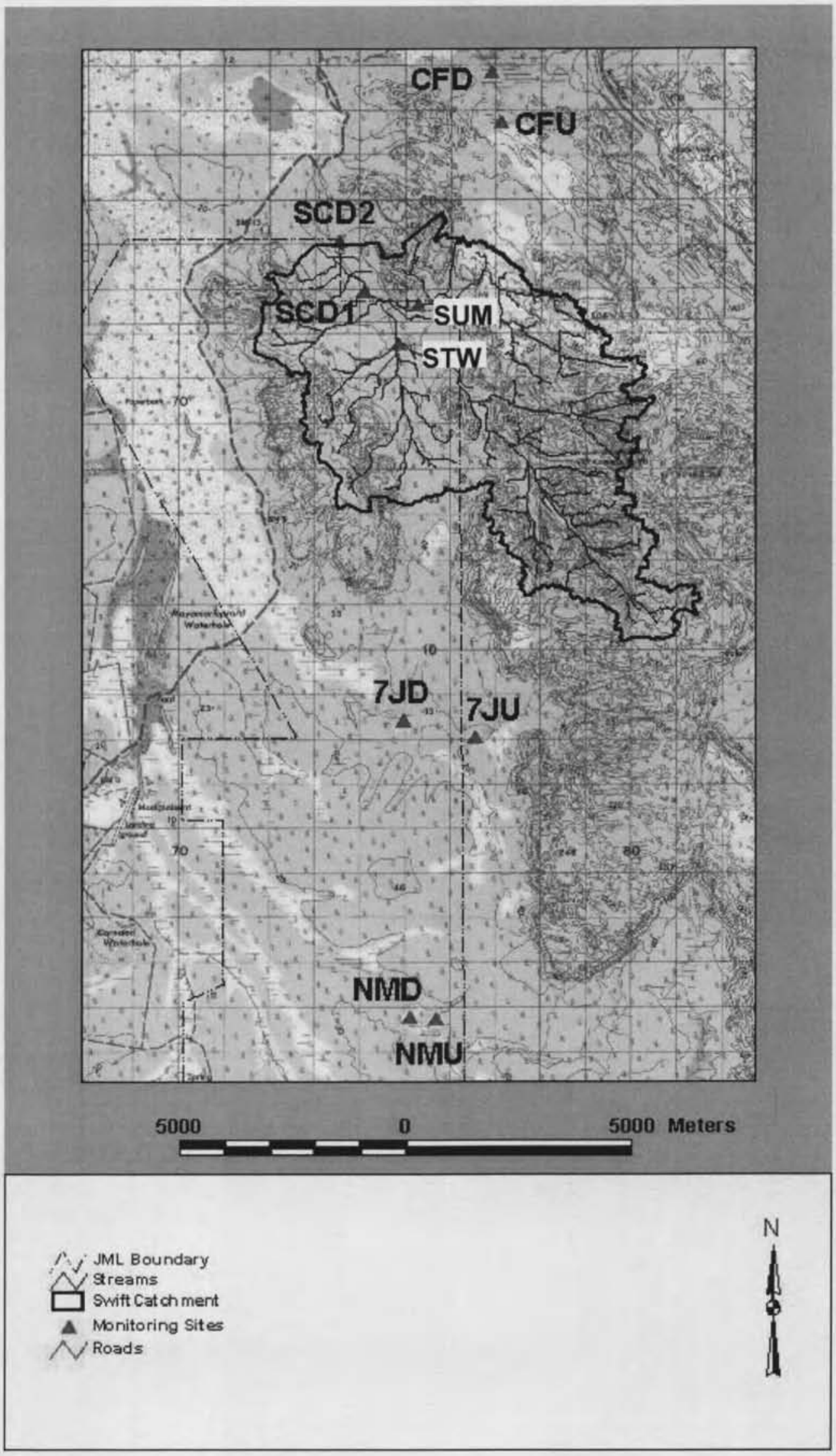


Figure 2.2 Map showing the location of the sampling sites for Swift Creek and the control streams - Catfish Creek, 7J Creek and North Magela Creek .

Erioocaulon spp, *Eleocharis* spp and *Cyperus* spp. The semi-aquatic grass *Pseudoraphis* sp, was also quite dominant along the edge of the creek banks, and was inundated for several of the sampling periods.

2.1.3 Description of main sampling sites

Paired upstream-downstream sampling sites were selected in accordance with broad requirements of the MBACIP design. The sites within each creek were chosen at approximately the same distance of 1 km apart (Fig 2.2). While paired sites were located in each stream, an additional downstream site was sampled at Swift Creek (1 km further downstream than the original downstream site) in order to detect any gradient effects from suspended sediment, if present. This would enhance inferences made and determine the extent of any possible impacts downstream in Swift Creek.

Apart from standardising the distance of separation of paired upstream-downstream sites, sites were also selected for their similarity of geo-physical attributes such as stream width, depth, flow patterns and also nature of riparian vegetation. Following the first sampling event, it was quite evident that Swift Creek West tributary, the upstream site selected in Swift Creek, differed physically to the downstream sites in Swift Creek. This site was flat and braided with negligible bank definition, largely dominated by *Melaleuca* spp and *Pandanus spiralis*, very unlike the deeply incised channels of the two downstream sites. As a consequence, an additional site - "Swift upstream main"- was sampled on each of the remaining sampling occasions. This upstream site represented a better physical and biological match for the downstream sites, being located on the main branch of Swift Creek, along a deep meandering channel dominated by gallery forest vegetation such as *Allosyncarpia ternata* and *Sygygium* spp. To ensure continuity, the Swift west tributary was also sampled on all remaining occasions to provide a further upstream comparison.

2.1.4 Location and brief description of main sampling sites

Sampling sites are shown in Figure 2.2 and in digital photos in Appendix 3. A brief description of the sites and their location follows.

Swift Creek

Swift Downstream 1 (code SD1) 8617919N 53274113E, located just upstream of the *eriss* and *ERA* gauging stations and immediately downstream of the entry of North tributary (The tributaries are described further on in section 2.1.6). This site was situated on a wide straight channel with well defined bank edges. Vegetation is dominated by *Pandanus spiralis* and and *Melaleuca* spp.

Swift Downstream 2 (SD2) 8619021N 53273578E, Located a further 1 km downstream of SD1 along a narrow channel with sloping bank edges and a deep pool located on a nearby bend. This site had a number of submerged fallen trees.

Swift Upstream Tributary West (STW) 8616759E 53274869E, Located 1 km upstream from SD1, and just upstream of a permanent billabong and the entrance of South tributary. This site was flat and braided with negligible bank defintion, largely dominated by *Melaleuca* spp and *Pandanus spiralis.*

Swift Upstream Main (SUM) 8616832N 532755224E, Located 600 m SE of SUM on a deeply channelled, meandering section of stream. Evidence of considerable recent natural erosion was observed during sampling, with several fallen trees and scoured banks present. Dominant vegetation here was gallery forest including *Allosyncarpia ternata* and *Sygygium* spp. This site was approximately 200 m downstream of the ***eriss*** gauging station.

Catfish Creek

Catfish Downstream (CFD) 8622783N 53276924E, Located 150 m upstream of a billabong on a wide channel (12 m width) of stream. Vegetation was fairly open with grasses dominating and a sparse overstorey of *Melaleuca* and *Barringtonia acutangula.*

Catfish Upstream (CFU) 8621684N 53277153E, Located along a straight stretch of stream approximately 150 m downstream of a small waterfall exiting the edge of the sandstone escarpment. Bank edges were steep with abundant macrophytes. Vegetation consisted of *Pandanus aquaticus*, *Syzygium* sp and *Melaleuca* spp.

7J Creek

7J Downstream (7JD) 8608394N 53274973E, Located approximately 300 m downstream of the ERA gauging station, before the confluence of the northern and southern arms of the creek. Channel cross-section was wide (10-12 m) and deep. Vegetation was dominated by grasses, *Pandanus spiralis* and *Melaleuca* spp.

7J Upstream (7JU) 8608039N 53276583E, Located on a bend in the stream, approximately 100 m below a small riffle area. The freshwater mangrove, *Barringtonia acutangula*, was located at this site, as well as the common *Pandanus spiralis* and *Melaleuca* spp.

North Magela

North Magela Downstream (NMD) 8601740N 53275106E, Located on a horseshoe bend of the stream, approximately 200 m downstream of the ERA gauging station. The channel was wide (10 m) with little overhanging vegetation, and contained several deep pools. Vegetation was generally sparse, apart from several mature *Syzygium* and *Melaleuca* spp trees, which formed a dense canopy immediately upstream of the sampling site.

North Magela Upstream (NMU) 8601687N 53275688E, Located on a straight stretch of stream with a wide, deep channel. The banks were mostly lined with a mixture of *Melaleuca* spp and *Acacia* spp. This site contained a submerged rocky ridge on the southern bank.

The codes for each site provided in the descriptions above, are used here-on when referring to individual sites.

2.1.5 Sampling frequency

The onset of wet season flow in Swift Creek commenced around the 10th of December and sampling commenced at all sites 10 - 12 days later. This lag period served two purposes: i) to allow debris and water of naturally-poor quality to pass through the system (Humphrey et al 1990), and ii) to allow recolonisation of macroinvertebrate fauna. Hereafter, sampling was conducted as closely as possible to a three-weekly time frame, continuing through to cessation of flow in early May. On each sampling occasion all sites were sampled over a period of three consecutive days. The sampling dates are provided in Table 2.1.

The three-weekly sampling regime would provide some independence of temporal *difference* (or dissimilarity) values, as this is a basic assumption of BACIP designs. Sampling at too frequent an interval has the potential to re-sample the same cohort of organisms thereby introducing trends into the data. The life cycles of most of the taxa would be completed within the 3-weekly period (see Marchant 1982a).

2.1.6 Tributary sampling sites

The Jabiluka mine site lies approximately 1.5 km to the east of the Oenpelli road, and is located between two prominent sandstone outliers (Fig 2.3). The gradient of the construction site is slightly inclined and soils comprise earths or texture contrast soils on colluvium, or deeper alluvia that form the floodplains and creekbeds of the drainage system (Kinhill and ERAES 1996). Construction has disturbed an area of 31.5 hectares, consisting largely of a run-off pond, hard-stand area and rock stockpiles. Swift Creek drains the mine construction site by way of 3 minor tributaries: North, Central and, to a far lesser degree, South tributary. The potential to detect a gradient effect of disturbance

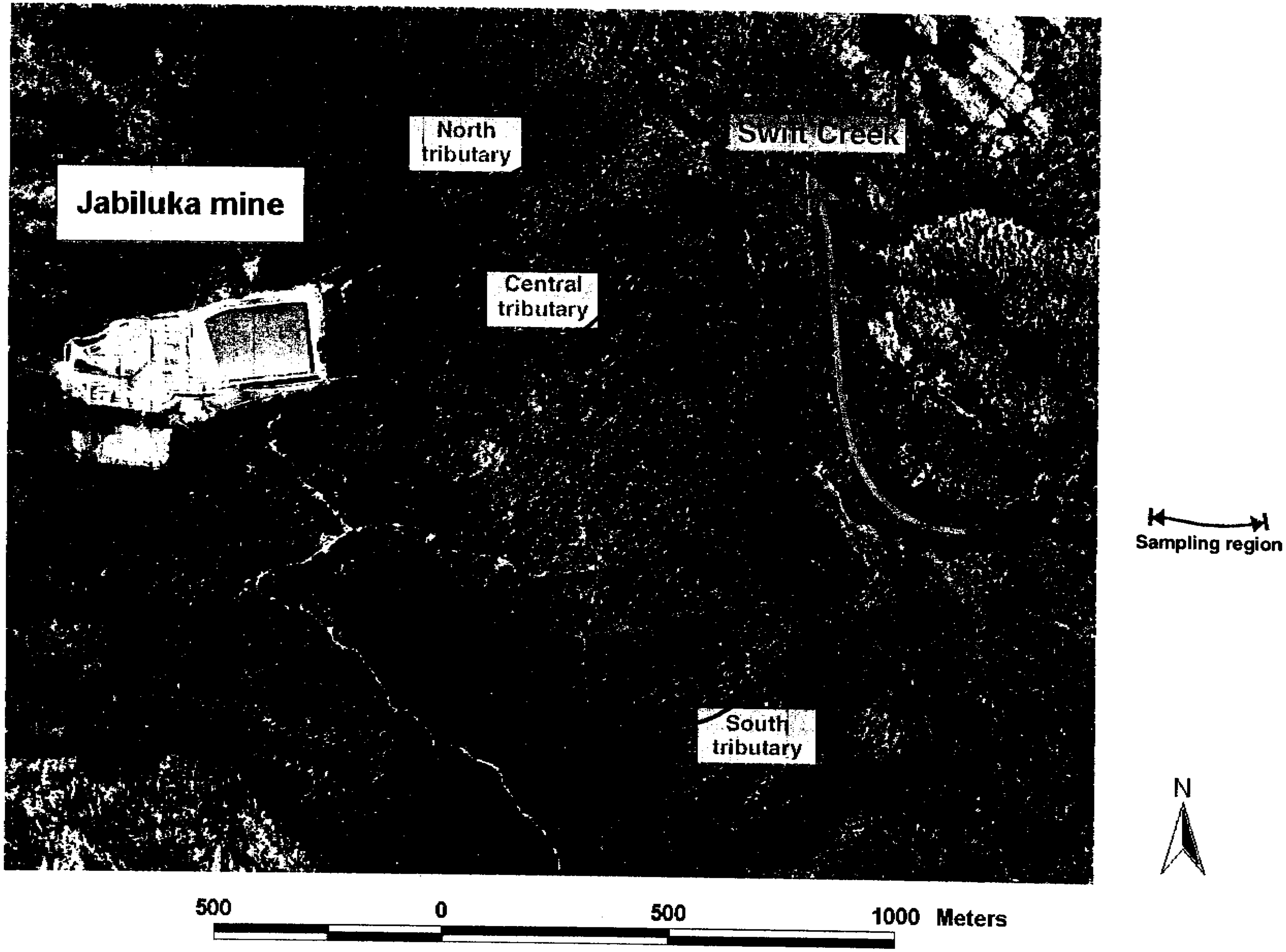


Figure 2.3 Map of tributaries North, Central and South, draining the Jabiluka mine-site

was available by way of late wet season sampling of sites located along each of these tributaries.

The tributaries are narrower than the main Swift Creek channels, yet they contain comparable riparian vegetation. North tributary, which was diverted during mine construction, now flows past the mine along the northern boundary before entering an underground seep, about 100 m downstream of the mine. It then re-emerges as a defined sandy channel similar to the other tributaries. Central tributary flows past the mine site, adjacent to its southern boundary and South tributary commences flow approximately 250 m southeast of the mine, thereby receiving minimal (if any) influence from construction activities.

Sampling was undertaken along each of the tributaries to coincide with the sixth sampling occasion in April. This recessional flow period is routinely sampled in ARR streams and billabongs by ***eriss*** for assessment of mine impact, as this timing represents summation of impacts occurring throughout the wet season, and because taxa richness and abundance of aquatic biota are high at this time (Humphrey et al 1990; Tripodi 1996). Sites were selected approximately 20-40 m apart along each of these tributaries between the mine site and tributary-Swift Creek confluence. Both North and Central tributaries produced six sites (1-6), whereas South tributary dissipated into the vegetation to the southern side of the mine, providing for sampling at only five sites (2-6). (Sites for South tributary were labelled from 6 down to 2 and for this reason there is no site 1).

The tributaries located close to the mine, North and Central, would provide the first indications, if any, of water quality deterioration arising from the mine. GPS locations for each of these sites are provided in Table 2.2. A further tributary, Swift tributary East (STE), entered Swift Creek on the eastern side (see Fig 3.2), opposite to the mine and was monitored by the Erosion and Hydrology sector of ***eriss*** for suspended sediment loads that may have had an influence on turbidity levels in Swift Creek.

Table 2.1 Sampling dates for all the creeks over the 1998/99 wet season

Sampling Event	Swift Creek	7J & North Magela	Catfish Creek
1	21/12/98	22/12/98	23/12/98
2	12/1/99	13/1/99	14/1/99
3	2/2/99	3/2/99	4/2/99
4	22/2/99	23/2/99	24/2/99
5	15/3/99	16/3/99	22/3/99
6	19/4/99	20/4/99	23/4/99
7	10/5/99	11/5/99	12/5/99

Table 2.2 GPS Locations for the Tributaries sampled on the 26-28th April

Tributary Site	GPS Location (North)	GPS location (East)
North 1	8617650	53273777
North 2	8617673	53273828
North 3	8617751	53273909
North 4	8617769	53274039
North 5	8617839	53274199
North 6	8617834	53274242
Central 1	8617307	53273673
Central 2	8617381	53273922
Central 3	8617421	53274015
Central 4	8617418	53274294
Central 5	8617296	53274260
Central 6	8617399	53274571
South 2	8616605	53274390
South 3	8616598	53274387
South 4	8616619	53274458
South 5	8616678	53274614
South 6	8616752	53274640

2.2 Field procedures

2.2.1 Choice of sampling habitat

Benthic invertebrates occupy a range of different aquatic habitats within rivers and streams. Macrophytes provide a very important habitat for invertebrates through shelter, production of detritus and epiphytes, as recruitment sites for larvae, while some dipteran taxa such as *Simuliidae* and some *Chironomidae* rely directly upon macrophytes at the attached stages of their life cycles (Kaenel et al 1998). Only the macrophytic-edge habitat was sampled in this study as it represents a relatively stable habitat within the fluctuating wet season flow regime, where large flood spates can cause severe disruption to communities (Tripodi 1996). Studies, both within the ARR and elsewhere, have noted the greater abundances and diversity of macroinvertebrates found within this habitat, particularly in comparison to sand-bed habitat that dominates ARR streams, including the Jabiluka streams sampled in this study (O'Connor et al 1997; Popperl 1996; Tripodi 1996; Batterham 1999). Often, macrophyte species in the ARR are located amongst the root mat of riparian vegetation such as *Pandanus spiralis* and *Melaleuca* spp which could potentially add further to habitat diversity.

2.2.2 Sampling techniques

Macrophyte-edge habitat was sampled at each site and on each sampling occasion by slowly dragging a D-shaped pond net, 500-*u*m mesh size, along the edge of the substrate and agitating by hand the substrate surface and macrophytes to dislodge any macroinvertebrates. The direction of sampling was against the stream current so that when the macroinvertebrate fauna became suspended in the water column it was washed downstream into the pond net. This technique was repeated 3 times over a fixed 5 m transect for each replicate sample. Each replicate sample was 1.7 m^2 in size (0.34 m width of the pond net x 5 m length of creek sampled).

After collection, the sample was tipped into a 20 L bucket half filled with clean creek water. Coarse plant material was shaken, washed free of invertebrates then discarded. The contents of the bucket were then elutriated to suspend the lighter organic matter and invertebrates, which were then tipped onto a 500 *u*m sieve. The elutriation and sieving processes were repeated several times until the debris, which remained in the bucket, was free of macroinvertebrates and fine organic material. A coarse sort of remaining material in the bucket was carried out in the field to ensure all fauna had been removed. The organic material retained in the fine sieve was then transferred to a plastic jar containing 80% ethanol solution, for preservation and later laboratory processing.

2.2.3 Number of replicates collected

For community studies, the number of replicates to be collected at a site on any sampling occasion is often determined by the plateau reached in (i) taxa accretion curves, (ii) paired-site dissimilarity decrement curves, or (iii) specified error rates of mean organism abundance. Studies relevant to taxa accretion curves have indicated that between four and six replicates are needed before the rate of recording of new taxa declines (Marchant 1982b; Metzeling et al 1984; Doeg et al 1989). However, this is dependent upon the sampling area of the individual replicates. Marchant (1990) examined paired-site dissimilarities and found that less than 4 replicates did not adequately describe the patterns that were found in a full data set. In a two year study, Resh and McElravy (1993) suggest that for several benthic measures, increasing the number of replicates above 5 did not reduce the error rates between means significantly. While Green (1979) recommended a minimum of three replicates for general comparisons of community structure, collectively, these studies would suggest that more than four replicates may be needed to adequately describe the community structure present.

In the ARR, sample size has been found to vary depending upon the nature of the stream and the season. Thus, under late dry season conditions in riffle habitat of the upper South Alligator River, where flow is permanent, four x 0.06 m^2 replicates were adequate to characterise community structure at a site (Faith et al 1995) whereas in the early dry season in seasonally-flowing Magela Creek where habitat is more variable, five x 0.31 m^2 replicates are required (***eriss*** unpublished data). Given that the determination of sample size requires detailed sampling and analysis, this issue could not be evaluated for the small Jabiluka streams prior to this study. Instead, experience and professional judgement of other ARR investigators was used to best determine this aspect. A large sampling area per replicate was selected (1.7 m^2), but with a tradeoff of relatively few replicates (3) per site (C Humphrey, ***eriss***, pers. comm). To explore this aspect further, on the 6th sampling event in April, six replicates were collected at each of the sites in order to examine the patterns of taxa accretion and between-site dissimilarity with increasing replication.

2.2.4 Physico-chemical variables

At each site and on each sampling event, a water sample was collected in an acid-rinsed one-litre bottle for later analysis of water physico-chemistry. This sample was kept cold (on ice) until brought back to the laboratory for analysis. Basic parameters measured were alkalinity, bicarbonate, ortho- and total phosphate, total organic carbon, alkali metals (Na, K, Ca, Mg) and heavy metals (Al, Fe, Mn, Cu, Pb, Zn, U, Ni, Cd, Cr). Also at each site and on each sampling occasion,- pH, conductivity, dissolved oxygen, temperature and turbidity were measured *in situ* using either a portable Hydrolab or a Horiba water quality meter. Both instruments were laboratory calibrated prior to use and rechecked again after use in the field.

In streams of the ARR, turbidity is a highly-fluctuating variable during wet season flows; the general trend shows turbidity and suspended sediment concentrations correlating with the rising stage of high flow events (Hart et al 1982). Thus spot readings taken every 3 weeks with routine macroinvertebrate sampling are of limited value. For this reason, turbidity values were also obtained from automatic readings derived from a data logger and turbidity meter installed at the gauging stations along Swift Creek (data provided by Michael Saynor, ***eriss***) in order to better estimate turbidity levels at the upstream and downstream locations over the wet season period. The downstream gauging station was located just below SD1 and the upstream gauging station was situated approximately 200 m upstream of the SUM site.

Turbidity values were taken as a surrogate estimate of suspended sediment in the water column. Stowar (1997) measured both turbidity and suspended sediment during the dry season in Jim Jim Creek, (Kakadu National Park), and found a strong correlation between the actual suspended sediment loads measured in the creek and the surrogate turbidity readings. Other studies have also found that turbidity readings provide accurate estimates of suspended sediment loads (e.g. Lloyd et al 1987). The ease of measuring turbidity as opposed to suspended sediment favours this method in a short-term study where resources and time are limited. However, the potential limitations of using surrogate measures must also be considered.

Methods used to analyse the water chemistry variables are outlined in Appendix 4.

2.2.5 Environmental and habitat variables

At each site, information was recorded for the following variables:

- Sediment composition (visual estimate of percentage of clay, fine sand and coarse sand);
- Surface flow rate (time taken for a float to travel a distance of 2 meters);
- Water depth (average of 3 measurements taken along each 5 m transect)
- Macrophyte composition and abundance. A visual assessment was made along each 5 m transect and through the depth profile for total percentage macrophyte cover. The percentage occurence of each macrophyte species was also visually estimated and together these summed to the total percentage cover. To further investigate for relationships between gross morphological characteristics of the plants and invertebrate community structure, plant species were grouped according to the life form categories used by Sainty and Jacobs (1994), O'Connor et al (1995) and Corbett (1996). However, several of the macrophyte species were in different life stage forms than when described by the previous authors and were therefore placed according to the form and morphological characteristics observed in this study (see Table 2.3). This occurred because the habitats under study were distinctly different to those of a lentic body (the subject of study of the previous authors), with fast flowing currents occurring for much of the sampling period. Life forms in this study were therefore noted to be dominantly submerged rather than emergent, with leaf surface area of most of the plants being generally smaller than when found in a lentic waterbody. This was possibly due to the fast flowing currents, which would have made large surface areas difficult to maintain.

Table 2.3 Morphological groupings of plants found at the sampling sites over the wet season, as used by Sainty and Jacobs (1994).

Emergent Broad- leaf (EBL)	**Emergent Narrow- leaf (ENL)**	**Submerged and Emergent feathery (SEF)**	**Submerged-Not feathery (SNF)**	**Not feathery (NotF)**
Nymphoides	*Cyperus* sp. *Fimbristylis* Unidentified sp. *Pseudoraphis*	*Eleocharis*, *Eriocaulon caespitosissima*	*Limnophila*, *Triglochin*, *Blyxa*, *Isoetes* sp., *Xyris*, *Dysophylla*	Everything but SEF group

2.3 Laboratory procedures

2.3.1 Sub-sampling of taxa

Sub-sampling is an essential aspect of most benthic studies. Walsh (1997) established that for multivariate data analysis, it was more representative of the data set to sub-sample to a set number ("fixed-count" e.g. 200 animals) than a set portion (e.g.10%) of individuals. This is the practise currently used by the AUSRIVAS and MRHI programs. Humphrey and Thurtell (1997) found across a range of different habitats and regions of Australia, that a sub-sample of 200 animals consistently recovered at least 90% of the invertebrate families present at a site. Their results were used as the basis for sub-sampling to 200 animals in the present study.

For conservation assessment it is important to screen the sample for the inclusion of non-abundant (typically large body size) taxa-as described below. Non-abundant taxa also appear to be influential in determining relationships between environmental variation and community structure in large data sets (Faith and Norris 1989; Beisel et al 1998; Cao et al 1997).

2.3.2 Sub-sampling technique

Sub-sampling of each of the replicate samples was carried out using a 100-cell sorting box. (This consisted of a square box with 14.4 cm high walls and with 10 x 10 cells across the floor of the box, each cell separated by a thin metal vertical partition of 2.8 cm height). The contents of the replicate sample were firstly poured into a large jug full of water and mixed to evenly spread the sample. The contents of the jug were then poured evenly over the 100 cells of the sorting box. To ensure a complete and even spread, the apparatus was tipped up to about 30-45 degrees front and back, and side to side, so that eventually each cell "appeared" to contain approximately 1% of the entire sample (A description of the method is provided in Humphrey and Thurtell 1997). Two cells were then selected at random (using randomly-generated numbers) and their contents extracted (2% of the sample) using a suction device. The sub-sample was placed in ethanol to be sorted under a Zeiss stereomicroscope. Invertebrates were removed from this 2% sub-sample and the taxa and relative abundances counted in order to estimate the actual sample percentage required to achieve a count of 200 animals (for example, if 2% yielded 20 animals, then 20% would yield approximately 200 animals). Additional randomly-selected cells from the sub-sampler were then extracted to obtain enough material to sort to 200 animals.

A coarse screen was later undertaken of the remainder of the sample not sub-sampled and sorted by microscope. This residue was tipped into a large white tray and spread around with tap water so as to achieve an even spread. The sample was studied for 10 minutes, without microscope aids, and rare taxa visible to the naked eye (and not detected in the sub-sample) picked out. These taxa were identified and entered into data sheets separately from the sub-sampled component of the sample.

2.3.3 Taxonomic Identification

It would seem intuitive to assume that the highest (species) level of taxonomic resolution would always provide greater community description and detection of impacts. However, this is not always the case. Much seems to depend upon how the community is structured and whether or not several species dominate within Family or Order level. For example, an order or family dominated by just a few species would give a similar result at species or order/family level (Wright et al 1995). Stowar (1997) used family-level identification to successfully detect effects of turbidity downstream of a road crossing in

Kakadu. Other studies in the ARR have also used family level identification to detect mine-related impacts (e.g. O'Connor et al 1995, 1997).

More subtle impacts, however, may require species level identification for detection, and seasonal variation has been found to respond better at this taxonomic level (Marchant 1982b). Ideally, it would be desirable to obtain species-level data in any biological assessment program. Rarely, however, are resources available to do this; species-level identifications for example may require 3-4 times more resources than those required to identify to family level (C Humphrey, ***eriss***, pers comm).

In this study, macroinvertebrate fauna were identified to the family level with the exception of Oligochaetes and Nematodes that remained at the Class level, and water mites which were identified to either Order (Acarina) or Sub-Order (Oribatidae). The keys used for identification were Williams (1980), Hawking (1995) and unpublished keys specifically developed for the Alligator Rivers Region by taxonomic consultants to ***eriss***. These were supplemented by the ***eriss*** voucher collection.

2.3.4 Quality Control

Two operators were involved in sorting and taxonomic identification of the samples in this study. The author (GD) sorted 184 samples and a second person (MW) sorted the remaining 64 samples. To check for possible operator bias, a check and standardisation process was undertaken where 5 samples were cross-checked by the other sorter for correct identification and sorting effort (residues) in accordance with the procedures outlined in Hawking and O'Çonnor (1997). Briefly, this entailed a re-indentification of the identified taxa in each of the 5 samples by the opposing sorter, together with rechecking of the residue for any uncollected taxa. The two re-identified lists were compared to the original lists using the Bray-Curtis dissimilarity index. The residues were compared for percentage missed and therefore unidentified.

2.4 Data analysis

This study uses an MBACI(P) design, based upon the asymmetrical design of Underwood (1993), where three control (non-impact) streams are compared to one potentially impacted stream, providing a range of environmental variability within which to assess the ecological condition at the impact site, *post-impact*. As there are no *before* data and the analysis is reduced to *after* data gathered from the four streams; the inference is dependant upon dissimilarity values, together with paired site population abundance and taxa richness *difference* data, in Swift Creek that are significantly different from those at the Control Creeks (ANOVA). These formal hypothesis-testing procedures were complemented by Semi-strong hybrid multi-dimensional scaling (SSH-MDS), used to produce the multivariate ordinations. Inferences about impacts may be strengthened where "high" dissimilarity or *difference* values at the "impact" site are matched by the location of these sites in ordination space outside of the ellipse represented by the other sites. Inferences may be further enhanced if disturbance measures (e.g. turbidity) are found to correlate with the ordination space, as described in the following section.

2.4.1 Multivariate MBACIP dissimilarity values

The statistical analysis inherent within the MBACIP design relies upon dissimilarity values generated between upstream and downstream sites within independent control and potentially impacted streams. Paired dissimilarity values were therefore obtained between the upstream and downstream sites for the three control creeks: 7JU v 7JD, NMU v NMD, CFU v CFD and for Swift Creek using both upstream locations: SUM v SD1 and STW v SD1. A spatial gradient was examined using the paired dissimilarity values generated between SUM v SD2 only. Dissimilarity values were calculated using the Bray-Curtis index from the statistical package PATN (Belbin, 1993). Data were transformed by $\log_{10}(x+1)$ in order to normalise variance. Taxa that occurred only once throughout the entire data set were considered rare and deleted from the analysis as they can potentially distort the data (Gauch 1982).

2.4.2 ANOVA for MBACIP data

Analysis of Variance (ANOVA) tests the null hypothesis that population means of all samples are the same (Snedecor and Cochran 1989). Within the asymmetrical MBACIP design, ANOVA can be produced from (i) Bray-Curtis dissimilarity values (as described above), (ii) total taxa abundance *difference* data (upstream-downstream abundance) and (iii) total taxa richness *difference* data (upstream-downstream richness). Underwood

(1993) stresses the importance of looking at selective taxa that may show responses to particular disturbances and for this reason, univariate analysis was also undertaken on several taxa (Chironomidae, Leptophlebiidae and Simuliidae) which are known to be sensitive to elevated levels of suspended sediment (see section 1.2 and Appendices 1 and 2). These values were used within an ANOVA to test for two-way (Creek versus Time) interaction in the *after* period between impact and control creeks. This was done using a randomised combination of upstream and downstream replicates, as described below. All ANOVAs were performed using the statistical software package, STATISTICA (MathStat 1995).

2.4.2.1. Process for combining the upstream and downstream replicates

The MBACIP design relies upon *difference* values between up and downstream sites. In order to be able to obtain these values, the replicates taken from upstream and downstream must be combined. There are six potential ways to combine the sets of three replicates so that no auto-correlation occurs in the data (K McGuiness, NTU, pers comm). To do this, each of the six combinations was assigned a number (1-6) and a random number list generated so as to achieve a randomised selection of combinations. For example, number 1 may represent the combination of replicate 1 D/S versus replicate 1 U/S, replicate 2 D/S versus replicate 2 U/S and replicate 3 D/S versus replicate 3 U/S. Each number 1 brought about this particular combination of replicates. Numbers 2-6 had different patterns of combining the upstream and downstream replicates. In this manner, a randomly assigned combination was obtained each time the replicates were combined. This process was undertaken when combining replicates for dissimilarity, and for abundance and taxa richness *difference* data, and for the selected taxa.

2.4.2.2 Time series of paired upstream and downstream dissimilarities

Paired site dissimilarities for each creek were also examined in a time series plot over the wet season to assess for any anomalies within the pattern displayed by the impact creek (Swift), in relation to the control creeks, for this post-impact period.

2.4.3 Multivariate ordinations

Dissimilarity values generated between all sites were analysed in ordination space to (i) summarise temporal patterns of macroinvertebrate recolonisation, (ii) compare the ecological "position" of the two Swift downstream sites to all other sites over the sampling period and (iii) produce "snapshots" in temporal space for each of the sampling occasions, including the sixth sampling occasion which combined information from the tributaries draining the mine site (potentially indicating a gradient of disturbance).

Ordinations were produced using the SSH-MDS component of PATN software program (Belbin 1993). Through the PCC (Principle Axis Coordination) analysis, available in PATN, significant influential variables, both environmental and of individual taxa, of the ordination space can be determined. A further MCAO (Monte Carlo Analysis) analysis in this same statistical package is able to show the significance level of each variable in relation to the correlation. Significance levels were generated using 100 random starts in order to show $p < 0.01$ as significant (Belbin 1993).

Chapter 3: Results

3.1 Environmental Data

3.1.1 Turbidity

Rainfall in the Swift Creek catchment was erratic and, at times, intense, over the 1998/99 wet season, as can be seen from Figure 3.1. Rainfall commenced in the catchment in early December, and showed a declining trend throughout the month of April to cease later in the month. Flow continued for several more weeks into May. The movement of suspended sediment has been found to correlate to the rising stages of flood events (Hart et al 1982), and therefore turbidity readings would be expected to fluctuate over the wet season accordingly.

Relatively high turbidity was recorded for all creeks on the first sampling event in late December, with the exception of Catfish Creek, SUM and SD2 (see Tables 3.2 and 3.3). Low levels of suspended sediment in Catfish Creek may relate to the more perennial nature of this stream, and its location close to the sandstone escarpment and headwaters. The early establishment of flow, potentially, had also moved sediment and accumulated dry season debris down the creek, well before the wet season flushes began for the other three streams. Turbidity levels declined in Swift Creek (SD1 and STW), 7J and North Magela, after the third sampling event, falling to within the range of 0 – 4 NTU by the recessional flow period. However, spot readings of turbidity do not capture trends and events that correlate with fluctuating discharge regimes over short time periods.

Turbidity measurements were taken regularly in Swift Creek by the Erosion and Hydrology program of ***eriss***, using automatic data loggers ("Greenspan Smart Sondes") operating at the SD1 site, the SUM site and at the Eastern tributary (STE). The downstream site (SD1) received run-off from both the mine and the eastern tributary (Fig 3.2). Daily, and often hourly readings were averaged over the week to standardise values so that a comparison could be made between the Swift upstream and downstream stations, as recording frequency did not coincide on a daily basis (Fig 3.3). To show the relative differences between D/S and U/S turbidity, U/S values for both sites (SUM and STE) were subtracted from D/S values, and values for each site,

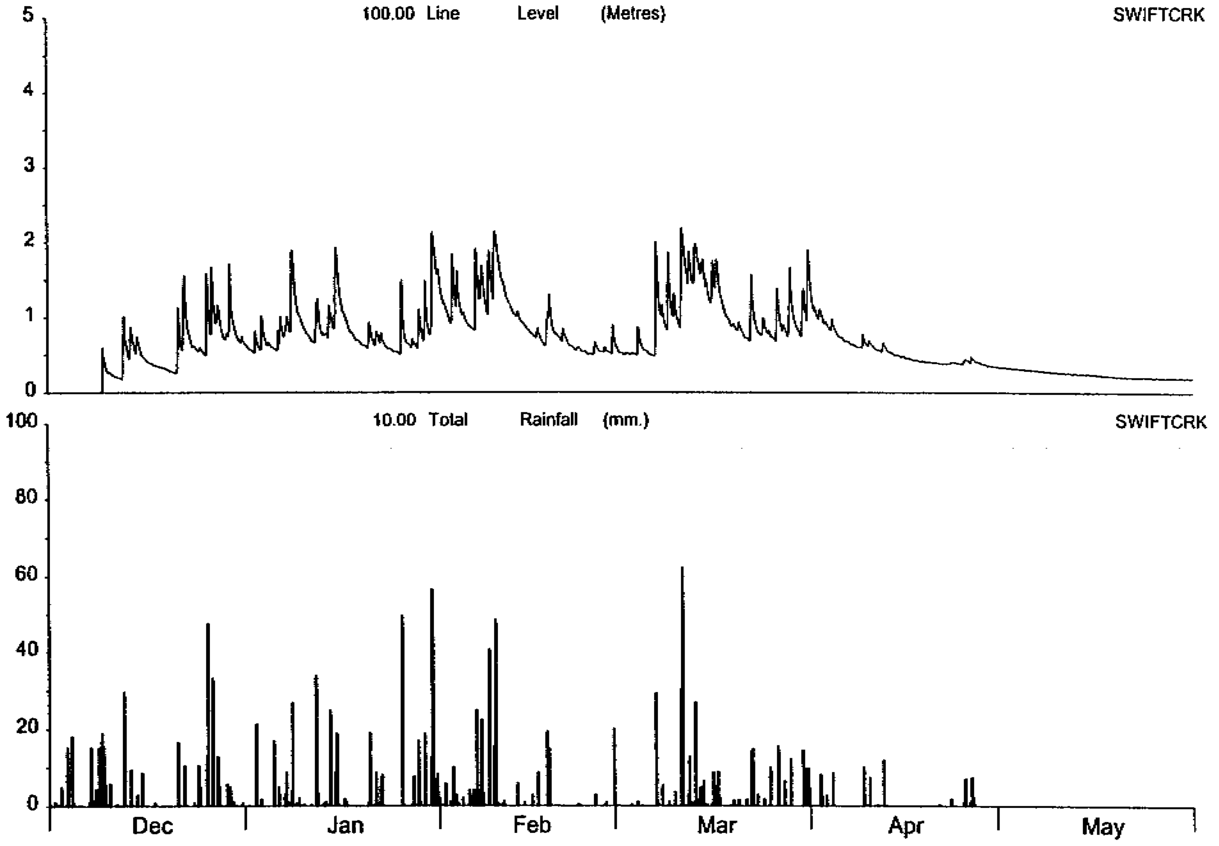
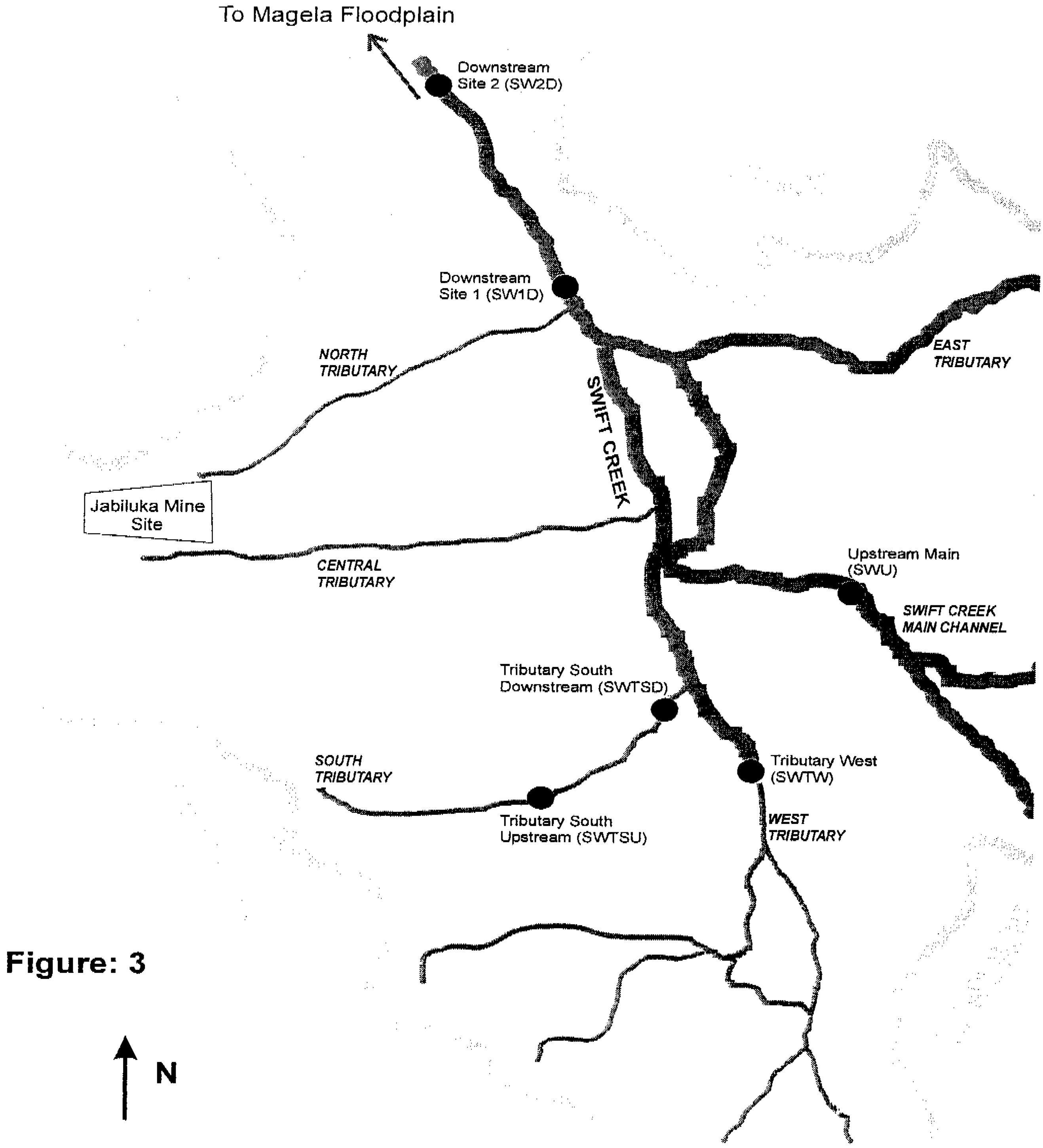


Figure 3.1 Plot of stage height and rainfall at Swift Creek (SD1 site) during the 1998-99 wet season.



EWL Sciences ABN 12 000 955 171	**Title:** **Swift Creek Benthic Macroinvertebrate Sampling Sites** Registered Research Agency No 31639	Drawn: R.B.	10/8/00	Scale
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Figure 3.2 Swift Creek sampling sites, showing the Eastern tributary: STE (data supplied by EWLSciences).

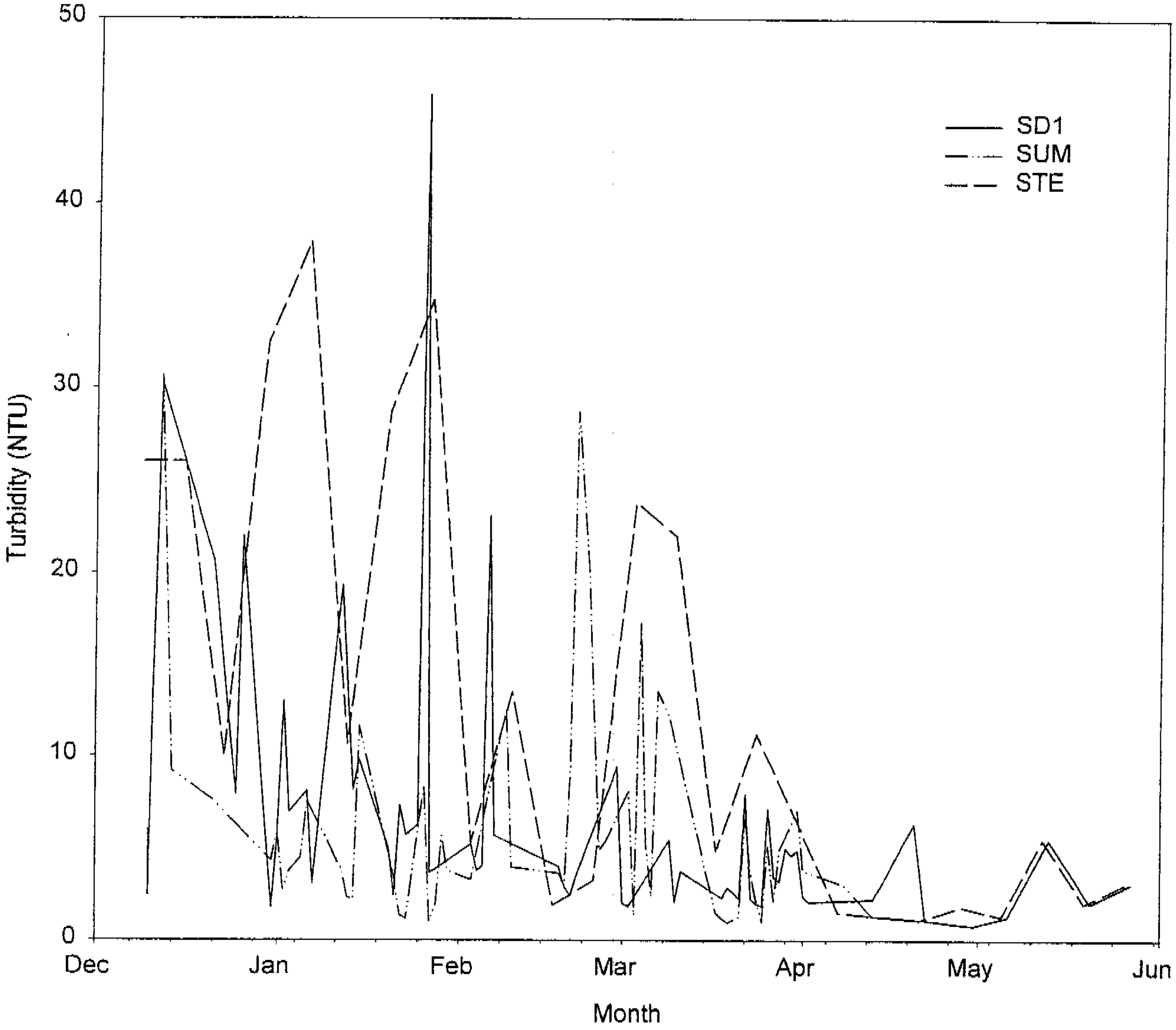


Figure 3.3 Turbidity levels (NTU) from three gauging stations in Swift Creek over the wet season: SD1, SUM and STE (Swift tributary east).

together with difference values, then plotted. If D/S values were consistently higher than U/S values (due to suspended sediment) we would expect to see the plotted lines on the graphs sitting well above the 0 line (Figs 3.4 and 3.5).

Figure 3.3 shows variable turbidity readings for all three Swift Creek stations with highest values at the STE site. Suspended sediment levels taper off to low values at all sites by April. The relative differences between SD1 and the two upstream locations are shown in Figures 3.4 and 3.5. The SD1 site had higher turbidity than the upstream SUM site for the early part of the wet season. The highest suspended sediment loads, however, were recorded for the Eastern tributary as can be clearly seen from Figure 3.5, where turbidity levels for SD1 are substantially less than the sediment loads recorded at the STE site. There is however, no information on discharge for STE and therefore it is difficult to determine what contribution this suspended sediment made to Swift Creek.

The maximum turbidity range for a single day recorded in Swift Creek at the downstream 1 site (SD1) was between 1 and 121.5 NTU, on 26.1.99. Values recorded at the ***eriss*** gauging station fluctuated markedly and the averaging out of these values over each week to make comparisons between sites masks any high values. Peaks in daily averages of 30 NTU (12/12/99), 21 NTU (21/12/99), 22 NTU (26/12/99), 13 NTU (2/1/99), 19 NTU (12/1/99), 64 NTU (26/1/99) and 23 NTU (6/2/99) were recorded. After this final date, turbidity readings were all under 10 NTU and generally under 5 NTU, indicating an easing of rainfall in the Swift Creek catchment (Fig 3.1). By contrast, turbidity was still noticeably high at STE over the month of March, predominantly during the first two weeks (Fig 3.3).

These comparisons illustrate the difficulty in interpreting data about mine influence on the downstream sites in Swift Creek, especially SD1. The potential exists for the Eastern tributary to have contributed high amounts of suspended sediment (as inferred from the turbidity readings) into Swift creek over the wet season, draining in above the SD1 site (see Fig 3.2).

The two main tributaries, North and Central, draining the mine site, (and on the opposite side to STE), both recorded high turbidity for a large part of December and all of January.

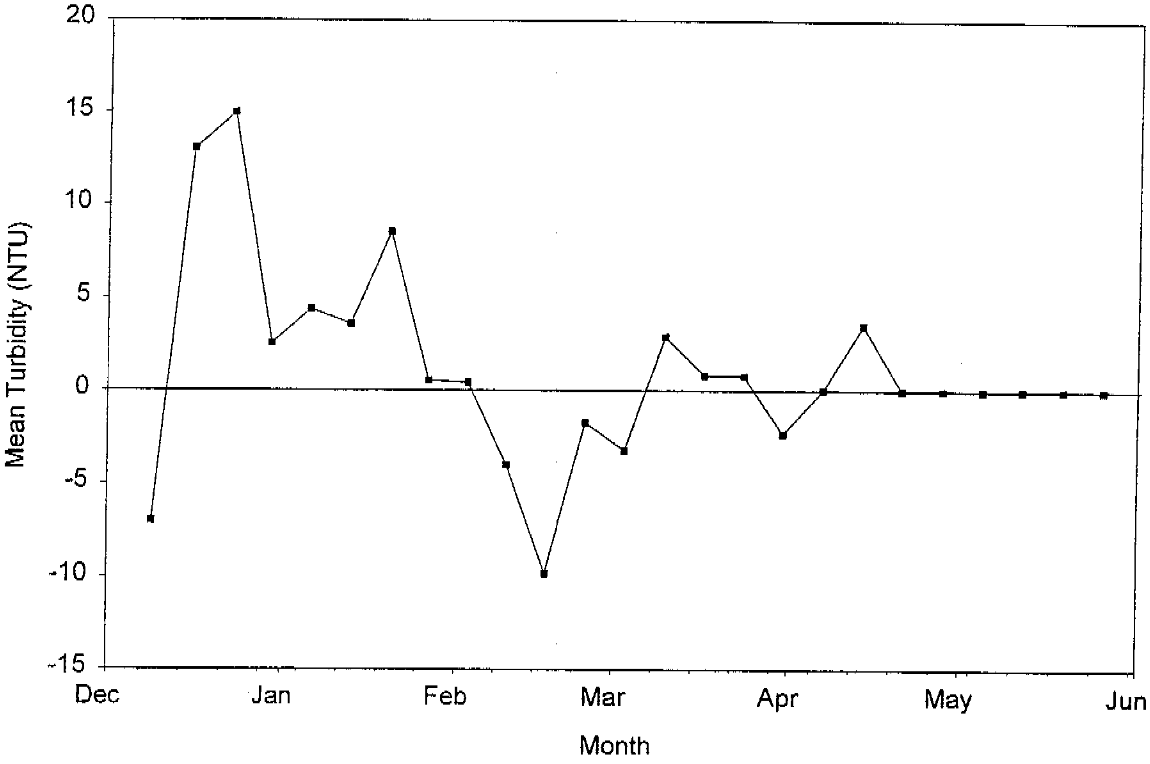


Figure 3.4 Mean turbidity in Swift Creek: SD1 - SUM

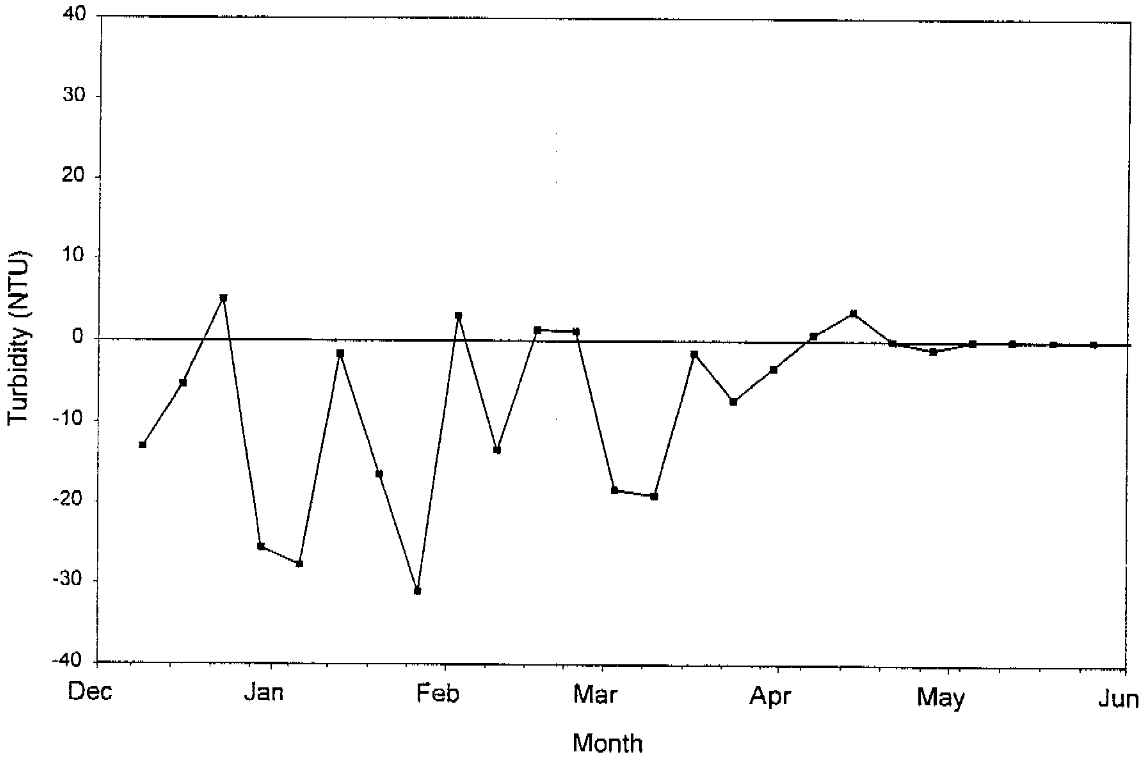


Figure 3.5 Mean turbidity in Swift Creek: SD1 - STE

This was most notable for the last two weeks of December where values were 3 and 4 times higher than the maximum values recorded by STE. It seems reasonable to postulate that these high levels were responsible for the elevated suspended sediment recorded at the SD1 site, as opposed to the SUM site, for the first 6 weeks of sampling. Average weekly values over the six weeks spanning mid-December to the end of January are shown in Table 3.1. It is clear from these results that the two main tributaries, North and Central, recorded far higher turbidity than the relatively unaffected South tributary, especially early on as flow recommenced. This indicates that mine influence was clearly a factor in the level of suspended sediment transported into these drainage streams, and potentially further down into Swift Creek.

Table 3.1 Average weekly turbidity levels (NTU) for the tributaries draining the mine site into Swift Creek

Date	North tributary	Central tributary	South tributary
14 – 20 Dec	133.9	77.0	5.6
21 – 27 Dec	40.1	161.8	5.1
28 – 3 Jan	17.4	61.6	4.6
4 – 10 Jan	18.1	31.6	3.5
11 – 17 Jan	21.2	27.2	3.6
18 – 24 Jan	31.5	NR	2.9
25 – 31 Jan	24.1	74.7	2.0

(Data provided by EWL S)

3.1.2 General water quality parameters

The water quality of the creeks in the Magela Creek catchment (Swift, 7J and North Magela) fell within a similar range, for the parameters measured (SUM was an exception for pH). Catfish Creek, in the East Alligator River catchment differed, from these streams for both pH and turbidity as described below. Results of the physico-chemical water quality analysis are presented in Tables 3.2 and 3.3 (and Appendix 5).

Catfish Creek recorded lower values for pH over the entire wet season (range of 4.16 - 5.0 at CFU and 4.07 - 4.9 at CFD). A t-test (paired 2 sample for means) was undertaken between the average pH value of the other creeks (less SUM because it was not sampled on the first sampling occasion) versus Catfish upstream and downstream locations. This analysis showed a highly significant difference between the Catfish and other stream sites (df=6, p<0.000 for both). This low range of values for both sites in Catfish Creek may be attributable to the close location and drainage through the sandstone massif of the Kakadu and Arnhem Land, which typically produces water of low pH (Humphrey et al 1990). The SUM site produced values within a similar range (3.97 - 5.4) to those found at Catfish Creek. The low pH of these sites over the wet season may also be attributable to decomposition of the considerable amounts of organic detritus from leaf litter found here, forming humic acids.

Conductivity fluctuated in creek waters over the wet season, with highest readings recorded early in the wet season, and then again during the recessional flow period. Values were low, typical of the Magela Creek catchment during a wet season (Hart et al 1982), ranges falling within 5 – 20 *u*S/cm. Conductivity values were only slightly higher than values recorded in flowing upper reaches of similar sandstone-draining Kakadu streams during the dry season by Stowar (1997). The low range of alkalinity values indicated the low buffering capacity of the waters (0.1 – 4.16 mg/L $CaC0_3$) which is again, typical of this sandstone region (Humphrey et al 1990)

Dissolved organic carbon was the main form of organic carbon recorded from the creeks and showed relatively high values compared to dry season data recorded by Stowar (1997). This accords with the greater amount of organic material being washed into the creek systems from surrounding watersheds during the early flood events of the wet season. While no values were recorded for the last two sampling events, a temporal trend of declining values is evident across the wet season.

Table 3.2 Average and range of pH, conductivity and turbidity for each site over the wet season

	pH		EC (*u*S/cm)		Turbidity (NTU)	
Site	**Average**	**Range**	**Average**	**Range**	**Average**	**Range**
STW	5.6	5.2-6.3	11.8	5.0-20.0	8.81	1.0-23.7
SD1	5.14	4.8-5.75	10.14	6.0-13.0	8.56	1.0-44.0
SD2	5.02	4.5-5.72	9.9	6.0-14.0	2.98	2.0-6.0
SUM	4.59	3.97-5.4	11.2	7.0-13.5	1.68	0.63-6.0
NMU	6.13	5.45-6.4	14.3	11.0-17.0	21.39	2.0-113.0
NMD	6.13	5.7-6.3	14.6	12.0-17.5	12.03	1.9-48
7JU	5.3	4.7-5.9	9.74	7.0-14.5	5.17	1.0-11.9
~~**7JD**~~	~~5.53~~	~~4.8-6.14~~	~~9.57~~	~~7.0-15.0~~	~~6.35~~	~~1.53-17.9~~
CFU	4.5	4.16-5.0	11.13	8.0-15.5	0.83	0.15-1.45
CFD	4.5	4.07-4.9	11.1	8.7-14.0	1.79	0.43-2.7

Results for Total and ortho-Phosphate are not included in Table 3.3 as data were only obtained for the first two sampling events for Total P, and for the first three sampling events for ortho-P; therefore no real trends could be inferred from the data. The values obtained were all low and within the expected range for these waters (C. LeGras, ***eriss***, pers comm).

Table 3.3 Average and range of water quality data, across all creeks at each of the seven sampling events, over the wet season

Average & Range	Sampling 1	Sampling 2	Sampling 3	Sampling 4	Sampling 5	Sampling 6	Sampling 7
pH	5.1	5.2	4.95	5.2	5.05	5.3	5.8
Range	4.4-6.0	4.5-6.2	4.1-5.7	4.0-6.3	4.1-6.2	4.1-6.3	4.8-6.3
Alkalinity (mg/L $CaC0_3$)	1.1	1.05	1.3	0.1	1.1	1.2	NR
Range	0.1-3.5	0.1-3.7	0-4.16	0.1-0.1	0.1-3.94	0.1-3.1	NR
EC (*u*S/cm)	12.69	12.12	11.8	10	7.9	10.3	15.2
Range	9.0-20.0	10.0-16.0	8.2-17.5	8.0-14.0	5.0-12.0	7.0-11.5	13.0-18.5
Turbidity (NTU)	28.2	6.4	5.4	3.1	4.7	1.73	1.56
Range	0.43-113.0	0.58-23.0	1.05-17.9	1.0-7.0	0.9-7.0	0.05-4.0	0.4-2.2
TOC (mg/L)	5.78	3.7	3.4	2.6	3.3	NR	NR
Range	4.36-7.34	2.71-5.26	2.2-4.65	2.03-3.32	2.55-3.9	NR	NR
DOC (mg/L)	5.8	3.8	3.24	2.34	3.2	NR	NR
Range	4.3-7.5	2.9-5.4	2.2-4.3	1.8-2.9	2.8-3.65	NR	NR

3.1.3 Metals

The water quality results for the metals analysis are found in Appendix 6. These results indicated no anomalies in the data for either of the two Swift downstream sites. All values for Swift Creek sites fell within the range represented by the control streams. There was a slight temporal trend of increasing values for Fe at all sites and the decreasing for Mn and Al; however, the general trend was one of fluctuation. The ranges of metals presented here are considered very low and typical of the oligotrophic stream systems characteristic of the ARR (C. LeGras, ***eriss,*** pers comm).

3.2 Quality Control and Integrity of biological data

Before describing and comparing spatial and temporal patterns in macroinvertebrate community structure amongst sites (section 3.3), factors affecting macroinvertebrate data quality and integrity are first provided. Section 3.2.1 discusses sample replication while section 3.2.2 discusses aspects of laboratory quality control and assurance.

3.2.1 Number of replicate samples per site

At the sixth sampling event, six replicates were collected at each site (instead of three) and sorted to assess the percentage increase in new taxa. The full range of taxa recorded did not level off until the sixth replicate indicating a gradual increase of taxa with increasing replication. This general trend was reflected across all sites and can be seen in Figures 3.6 and 3.7 for Swift Creek. The average values across all sites, from three random selections of replicate order, show that three replicates collected between 78 and 86% of the taxa, with the average over all 8 sites being 83.8%. This is just slightly lower than the percentages found by Humphrey and O'Connor (1997) of 90% for 3 replicates. Four replicates collected between 86 and 94% of the total taxa, and five replicates between 94 and 99% of the total taxa found.

Randomising the 6 replicates of Swift Creek upstream (SUM) and downstream (SD1) sites, Bray-Curtis dissimilarity values were calculated for cumulative replicate number (e.g. rep 3 SUM vs rep 6 SD1; average of reps 1 and 4 SUM vs average of reps 2 and 5 SD1; etc). The results are plotted in Figure 3.8. The dissimilarity range of 0-1 indicates community structure (taxa composition and relative abundance) of two samples/sites where the value "0" represents an identical match, through to "1" representing no shared features in community structure. For 3 replicates, dissimilarity is low (~0.37). Significantly, the addition of an extra 3 samples results in an improvement in the dissimilarity index of only about 10% ([1 - 0.33/0.37] x 100). While no obvious plateau in the dissimilarity value is reached, subsequent investigations by ***eriss*** have shown that when 12 replicate samples per site are compared between paired sites of the Jabiluka streams, a plateau is reached after 6 replicate samples (C Humphrey, ***eriss***, pers comm). The dissimilarity value is the basis of multivariate analysis. In this study it forms the basis of ordinations and paired site comparisons of community structure through MBACIP analysis. Given that the sampling effort per site (3 replicates) is only about 10% short of the "true" measure of community structure for a site, this sampling effort is deemed adequate for describing and comparing spatial and temporal patterns in community structure amongst sites.

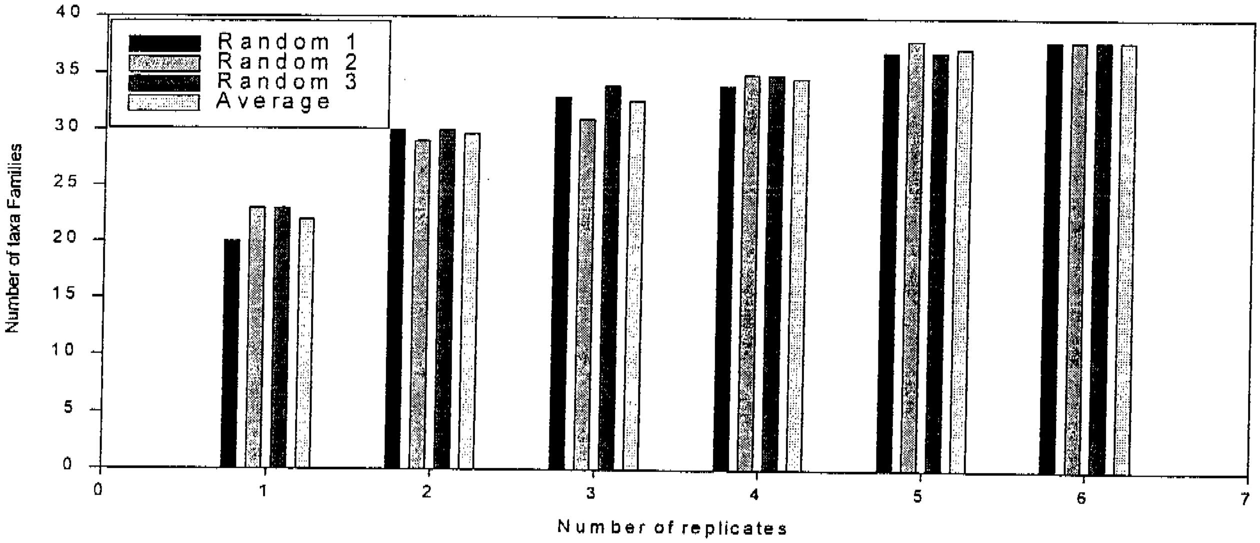


Figure 3.6 Taxa accretion over six replicates for Swift Creek, SD1 site.

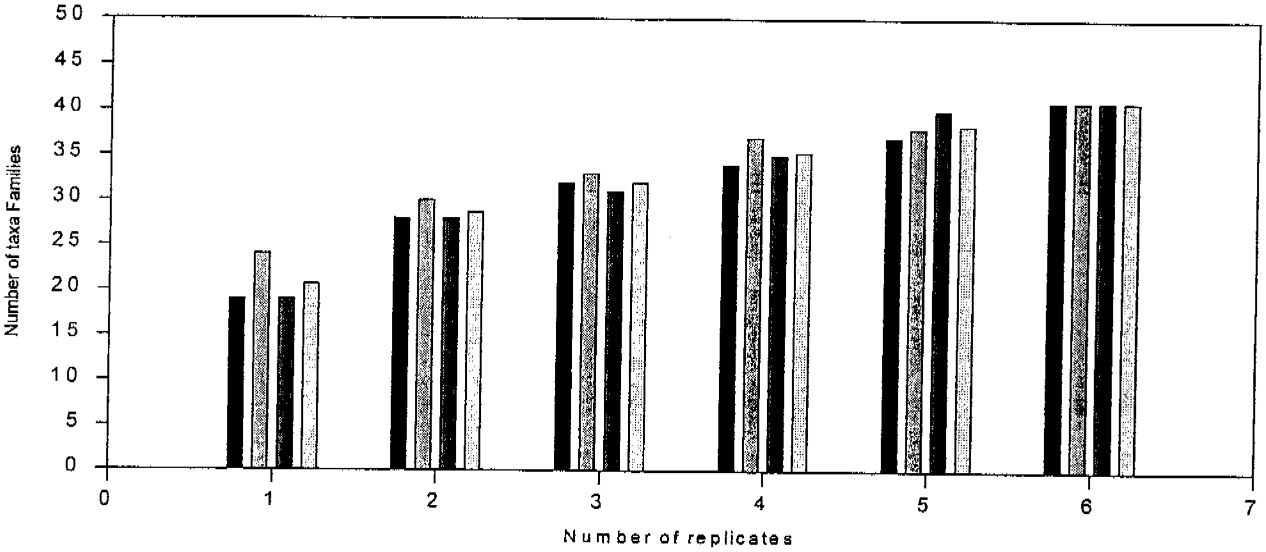


Figure 3.7 Taxa accretion over six replicates for swift Creek, SUM site.

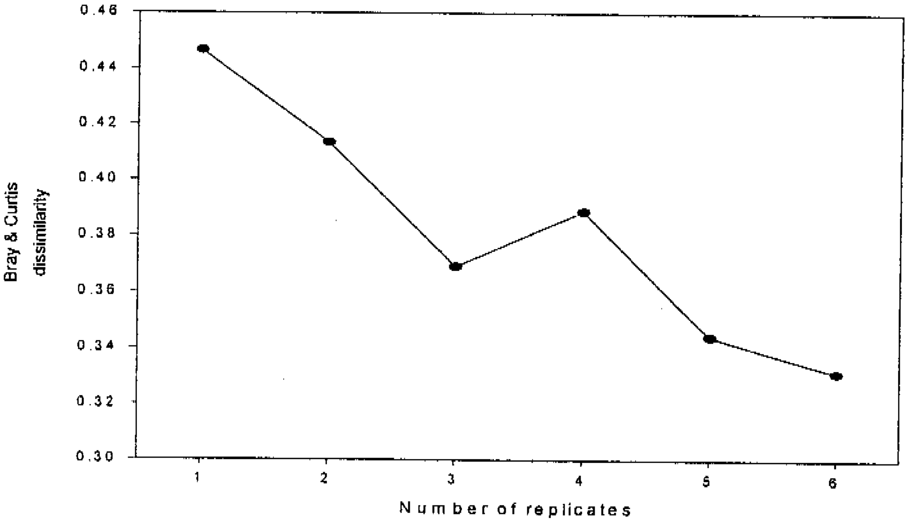


Figure 3.8 Dissimilarity between SUM and SD1 over six replicates collected at the sixth sampling

3.2.2 Quality control and assurance of taxonomic identifications and sorting efficiency

3.2.2.1 Identifications

Cross-checking of taxa identifications found no need for error corrections. The Bray-Curtis dissimilarity measure was calculated between the taxa identification lists for the two operators from five randomly selected samples. Each randomly compared taxa list produced a dissimilarity value of < 0.05, indicating a very close match between operators. However, a discrepancy was detected for Nematodes with the more experienced sorter (MW) picking up more individuals of this taxa than the author (GD). The necessary correction for this error is described in the following section (3.2.2.2).

3.2.2.2 Sorting efficiency

Cross-checking of residues highlighted an error between the more experienced sorter (MW) and the author (GD) for percentage of invertebrates remaining in the residue. GD missed several common taxa in the early samples and consistently missed Nematodes across all samples. To correct for this discrepancy, a 10% portion of all samples was resorted by GD. To ensure comparable data sets between MW and GD, samples were adjusted only where errors by GD exceeded the average error rate of MW. MW had an error range of 3 - 15%, averaging out to 10%. This is considered an acceptable range by established standards (eg AUSRIVAS: Hawking and O'Connor 1997, and in the U.S.A: Cuffney et al 1993).

3.3 Biological Data

3.3.1 Time series of paired dissimilarity values

Figure 3.9 showed a pattern of variable, yet similar, dissimilarity values between upstream and downstream sites for all creeks over the wet season. The two upstream locations in Swift Creek produced almost opposing dissimilarity plots, when each location was compared with SD1. The trend of greater dissimilarity values overall for SD1 v STW (the comparison of site SD1 and site STW), was to be expected as this site was notably different to the downstream sites as regards riparian vegetation, channel cross-section, depth and bank definition (see section 2.1.3). SD1 v SUM

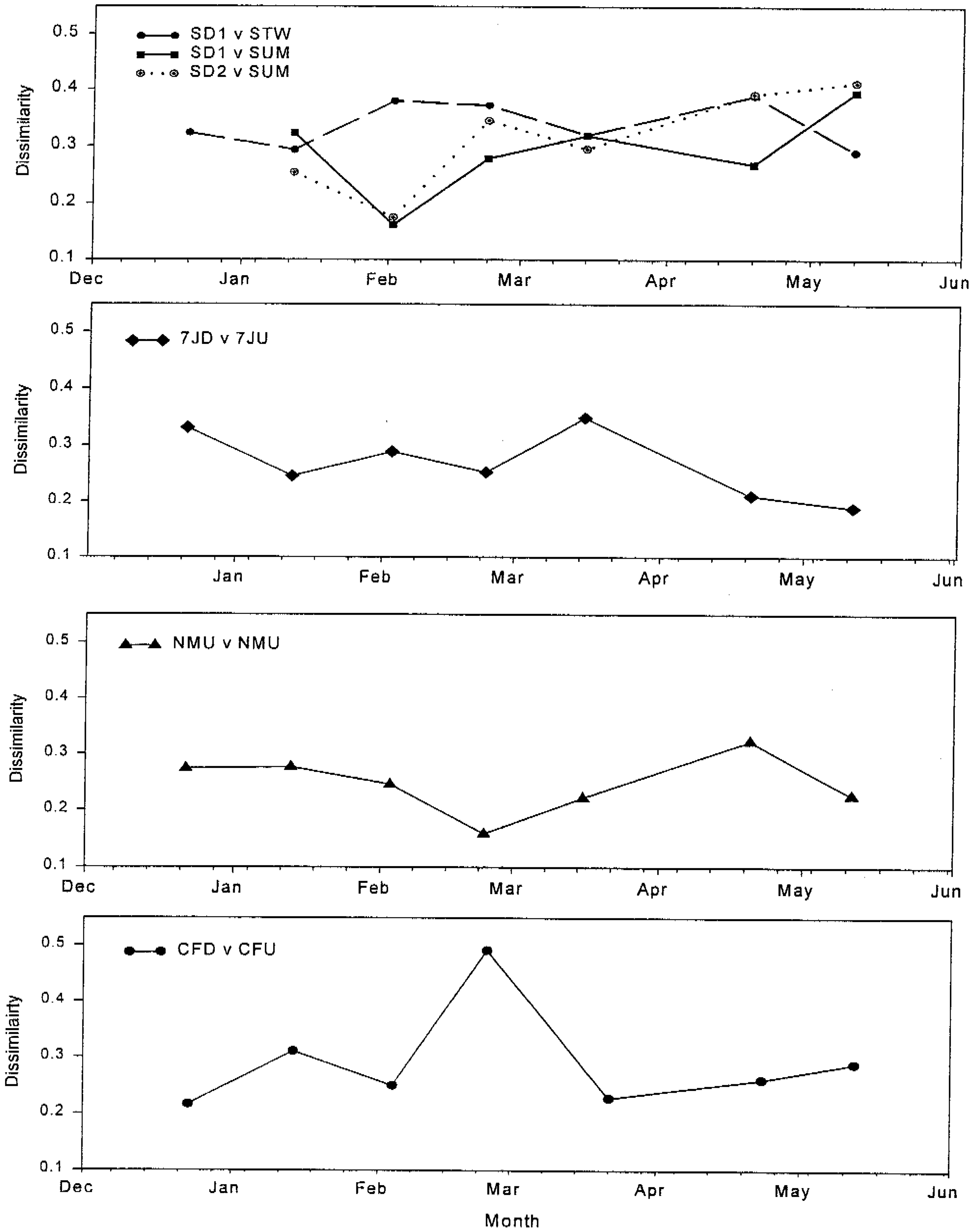


Figure 3.9 Multivariate dissimilarity between paired upstream and downstream sites in all streams over the wet season. Data has been log10(x+1) transformed.

showed fairly low dissimilarities overall, with the exception of the seventh sampling occasion.

7J upstream and downstream sites became "more similar" to one another by the end of the sampling period. Upstream and downstream sites in Catfish Creek were approximately similar over the sampling period, apart from the extremely low abundance in two of the replicates collected at the fourth sampling occasion at the downstream site. (A flood event in the East Alligator catchment caused a dramatic decline in abundance of macroinvertebrates at this site and the ponding of water over several days hindered recolonisation). North Magela showed a fluctuating trend over the wet season, but again, sites were approximately similar. No obvious temporal trends in the dissimilarity data were apparent. All control sites, and SD1 v SUM, fluctuated slightly over the wet season, between the values of approximately 0.2 and 0.35.

3.3.2 Statistical analysis of the data-ANOVA

In this study, a comparison was made between Creeks over Time, and was therefore a two-factor ANOVA. The MBACIP design used in this study relies upon measuring *differences* between upstream and downstream sites in order to be able to detect possible disturbance at the downstream site in the "potentially-impacted" creek (Faith et al 1995; Humphrey et al 1995). Therefore, the measures that were compared in this analysis were; paired-site dissimilarity, and *difference* (total abundance, richness, and abundance of selected taxon) data. Results of ANOVA show whether or not significant differences in means are found between any of the creeks overall, or if any significant interactions are produced between any of the creeks over the sampling period. Clearly, many interactions may be generated; however, of interest are those that compare only each particular sampling event.

3.3.2.1 Bray-Curtis dissimilarity

This analysis determined the statistical relationship of paired-site dissimilarity values between creeks over time (Fig 3.10). Table 3.4 shows that the main effect, comparing the means of the four creeks overall, did not produce any significant differences. However, significant interactions were evident ($p<0.03$) between the creeks at different sampling occasions. Further examination, based on a Tukey HSD test in the Statistica

(MathStat 1995) program, revealed that these significant interactions were not between the same sampling events, and therefore not relevant to the results. Therefore, Swift Creek and the three control streams did not differ statistically over the wet season, based on the measure of paired-site dissimilarity.

3.3.2.2 Total taxa abundance *difference* between all upstream and downstream sites

The most noticeable observation from the abundance *difference* analysis is the convergence of abundance values for the two Catfish Creek sites (Fig 3.11). Difference values in this creek eventually fell within the range presented by the other three creeks. As a consequence of higher abundance of taxa overall, Catfish Creek differed significantly from the other three streams (Table 3.4). A Tukey HSD test showed that means differed at the $p<0.0002$ significance level with 7J; $p<0.004$ with North Magela; and $p<0.0001$ with Swift Creek. North Magela and Swift creeks also differed significantly for means ($p<0.014$) but not with enough statistical power to show any significant interactions for any of the sampling occasions. This significant difference between these two creeks overall appears to relate to the low values recorded for Swift Creek over the final three sampling occasions where low abundance was found at the upstream main site relative to the downstream site (see Fig 3.22 and section 4.2 for discussion).

The Tukey HSD test highlighted several significant interactions between Catfish Creek and 7J (3rd sampling event; $p<0.02$, and 4th sampling event; $p<0.003$), and between Catfish Creek and Swift Creek (2nd sampling event; $p<0.04$, and 4th sampling event; $p<0.03$). As figure 3.11 indicates, the wide disparity in abundance *difference* data between 7J and Catfish Creek is evident at the 2nd and 3rd sampling events. The significant interaction between the paired-sites in Swift Creek and Catfish Creek was generated as a response to the large abundance found at the Catfish Creek upstream site early on in the wet season, relative to the Swift Creek sites at this time (see Figure 3.29). These results showed that Swift Creek fell within the range presented by two of the control streams, producing no significant interactions with North Magela or 7J for any of

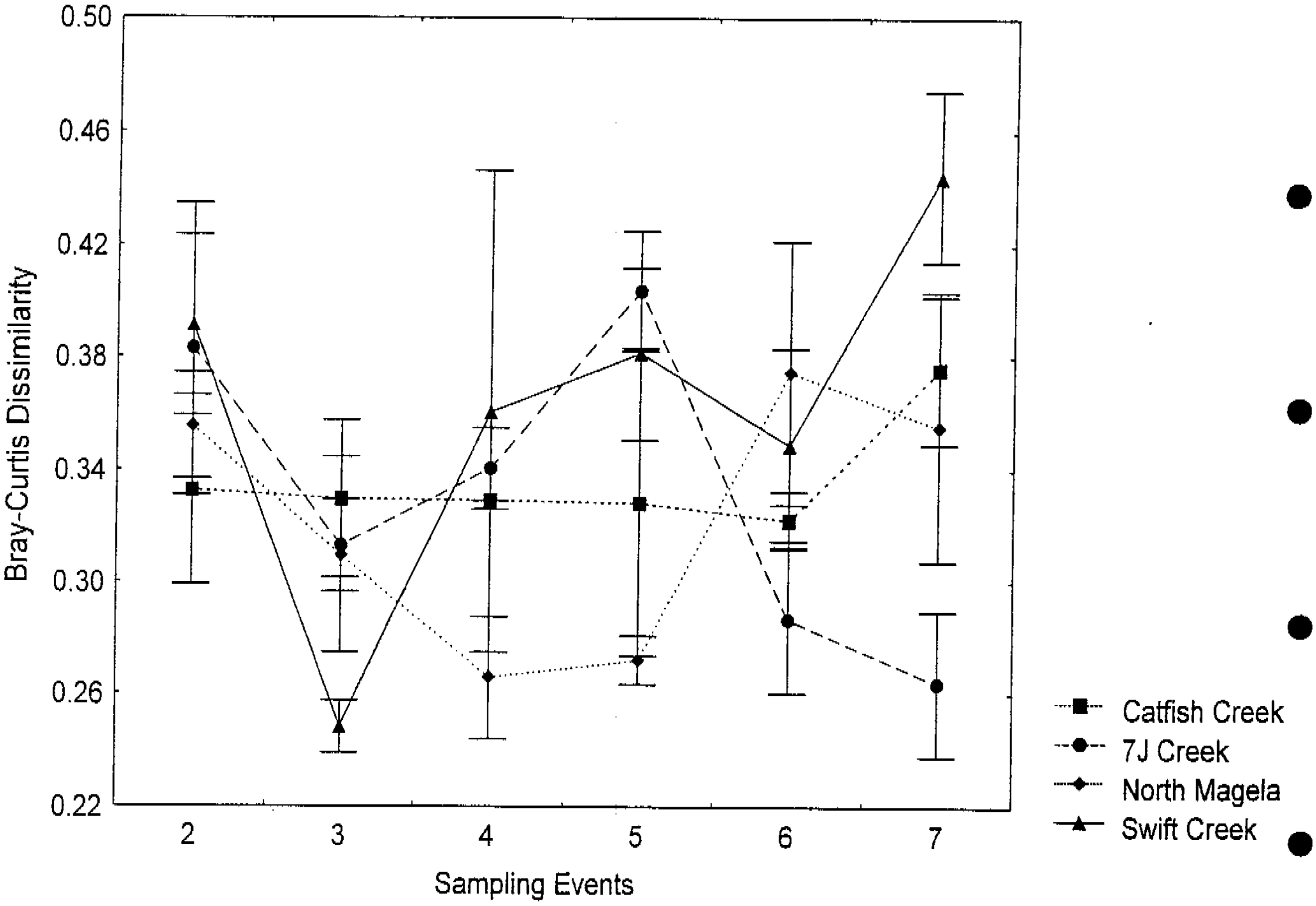


Figure 3.10 Paired-site dissimilarity values plotted over time for each of the creeks. Two-factor ANOVA showed F(15, 48)=2.05; p<.03. The Catfish Creek outlier (fourth sampling) has been excluded.

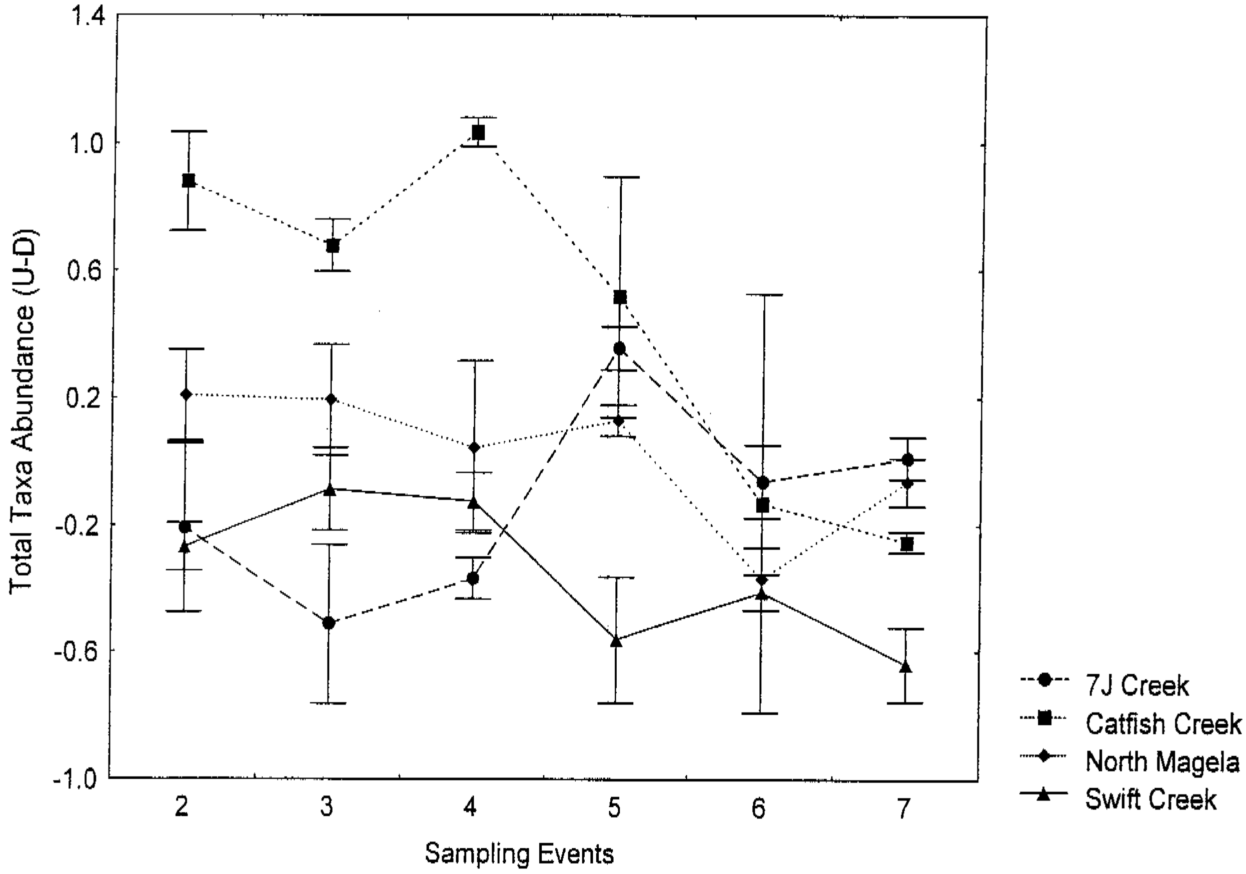


Figure 3.11 Total taxa abundance difference data (upstream - downstream) for all streams plotted over time. Two-factor ANOVA showed $F(15,48)=2.68$, $p<0.005$. Data has been $\log_{10}(x+1)$ transformed.

the sampling events.

3.3.2.3 Taxa richness *difference* between all upstream and downstream sites

Taxa richness was assessed using *difference* data between the upstream and downstream sites (Fig 3.12) and no significant differences between means were noted for any of the creeks (Table 3.4). Although results indicated significant interactions ($p<0.02$) between the creeks over time, a Tukey HSD test recorded no such interactions, indicating that statistical power was weaker within this test than within the ANOVA. All creeks showed a diverging trend, moving from values near to 0 at the beginning of the wet season (indicating similar taxa number at upstream and downstream sites), to more separated values by the recessional flow period. 7J Creek had higher taxa richness at the upstream site than the downstream site by the end of the sampling period.

3.3.2.4 Univariate analysis of selected taxa abundance

Three taxa (Chironomidae larvae, Simuliidae larvae and Leptophlebiidae) were selected from the total taxa data set for separate analysis as they were found to be significant within the ordinations in the direction of Swift Creek (a negative correlation for Chironomidae). These taxa have all shown responses to elevated levels of suspended sediment in previous studies (Stowar 1997, Davies and Nelson 1994, White and Gammon 1977). These taxa were examined using upstream-downstream difference data as undertaken in the previous section. Thus, differences in selected taxa abundance between the paired-sites in each stream were analysed within a two-factor ANOVA, examining Creeks over Time for any significant differences between means, or interactions. Leptophlebiidae abundance failed to satisfy the assumption of homogeneity of variances. As this can increase the probability of a Type I error (i.e. rejecting the null hypothesis when it is in fact true, and therefore there are no statistical differences between means), these results must be interpreted cautiously (Underwood 1997).

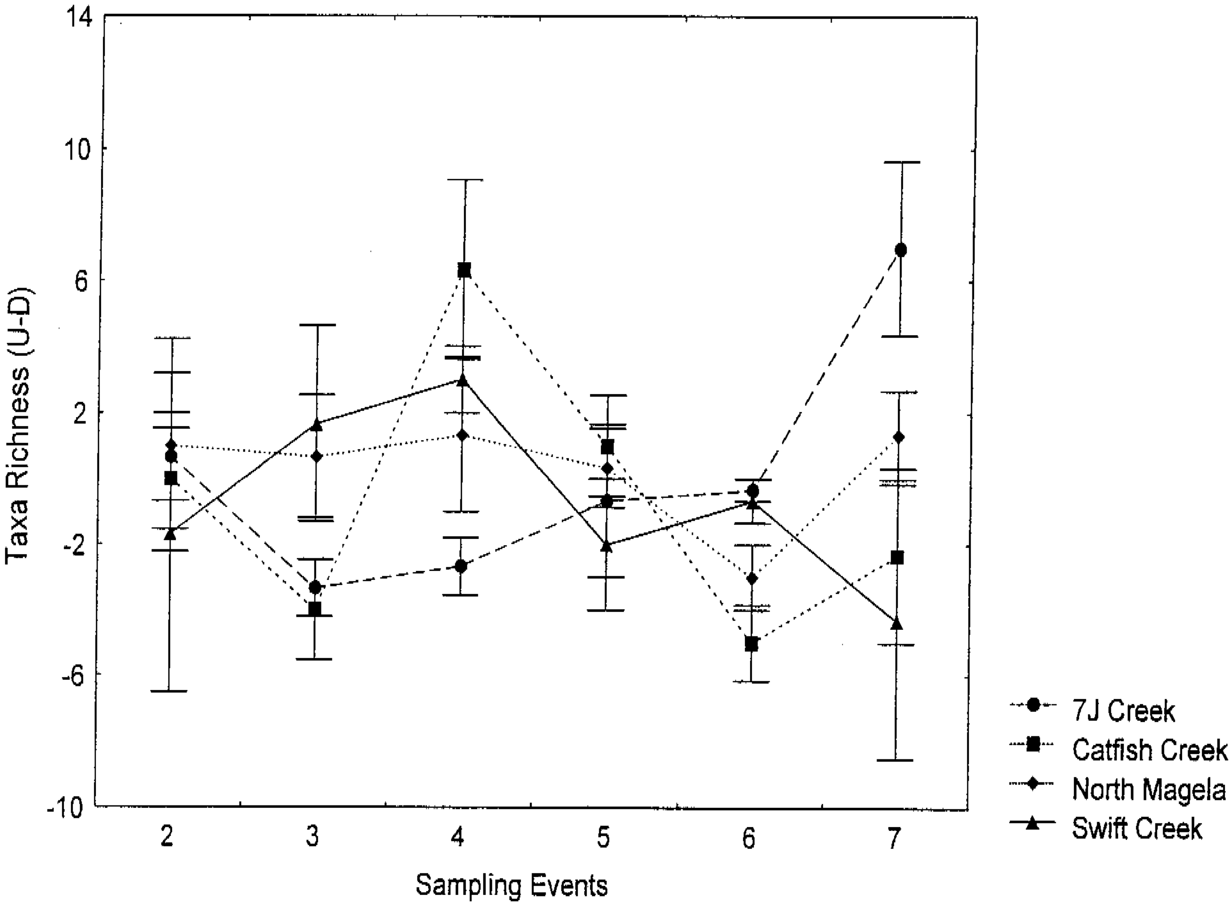


Figure 3.12 Taxa richness difference data (upstream -downstream) plotted over time for all streams. Two-factor ANOVA showed F (15,48)=2.17; p<0.02.

3.3.2.4.1 Abundance difference of Chironomidae

Table 3.4 shows that the means between the Creeks were significantly different. The Tukey HSD test demonstrated that Catfish Creek was again different to Swift Creek ($p<0.0005$), 7J ($p<0.0002$) and North Magela ($p<0.002$). Also North Magela and 7J differed significantly at $p<0.02$. Catfish Creek had far higher abundance of this taxon at the upstream site overall (see Appendix 8) and 7J had lower abundance at the upstream site early on in the wet season (see Fig 3.24). When sampling events were compared for interaction between Creeks, using the Tukey HSD test, 7J and Catfish were significant at the 3rd ($p<0.003$) and 4th ($p<0.0004$) sampling events respectively. Figure 3.13 graphically demonstrates the wide difference observed. Swift Creek and 7J also produced a significant interaction at the 4th sampling event ($p<0.004$) where low abundance of Chironomidae was recorded at the 7J downstream site.

This analysis showed Swift Creek falling within the range presented by the Control streams. Catfish Creek and 7J produced opposing trends for *difference* data between their upstream and downstream sites. The data demonstrated a convergence for all the creeks by the end of the sampling period, indicating that similar numbers of Chironomidae larvae were found at upstream and downstream sites in each stream, by the recessional flow period. Chironomidae are known to dominate during early, and also late, flow conditions (Paltridge et al 1997; Stowar 1997).

3.3.2.4.2 Abundance difference of Leptophlebiidae

Figure 3.14 can only be interpreted visually as the data did not satisfy the assumption of homogeneity as discussed earlier. Leptophlebiidae abundance showed a similar trend for Swift Creek and North Magela, where higher abundance was recorded at both downstream sites by the third and fourth sampling events respectively. 7J demonstrated the opposing trend of increasing abundance, for the upstream site, by the fourth sampling occasion. No Leptophlebiidae were observed at either of the Catfish Creek sites over the sampling period. While it is not possible to draw any firm conclusions from this graph, nevertheless, it appears that Swift Creek demonstrated similar trends to North Magela Creek.

3.3.2.4.3 Abundance difference of Simuliidae

Table 3.4 indicates that a significant difference in means was found between the creeks overall, but upon further examination with the Tukey HSD test, these data were not statistically powerful enough to indicate which creeks were responsible (Fig 3.15). Only one significant interaction between Creeks over Time occurred, and this was between Swift Creek and North Magela at the 6th sampling occasion ($p<0.005$). Figure 3.15 describes the opposing trends that were produced between these creeks at this time. Swift Creek recorded higher abundance of Simuliidae at the downstream site relative to upstream, whereas, North Magela had higher abundance at the upstream site. This taxon was not observed at Catfish Creek, except in low numbers (an average of four individuals at the upstream site) during the second sampling event. It must be noted that this analysis does not include data from the first sampling event (because SUM was not sampled at this time), when high numbers of Simuliidae were found at both Swift downstream locations. This is discussed further in the following section 3.3.3

3.3.3 Trends in abundance of Simuliidae over the wet season

In order to best demonstrate the abundance patterns of Simuliidae, a three-factor ANOVA was used to compare not only Creeks over Time, but also the upstream and downstream sites (U/D). This data was therefore, not *difference* data, but abundance data for each site (Fig 3.16). Simuliidae abundance (data has been $\log_{10}(x+1)$ transformed) showed similar trends at the upstream and downstream sites in the Control streams. However, a rapid decline in population abundance is noted at SD1 after the first sampling occasion in December. The three replicates at the SD1 site recorded 90, 62 and 50 individuals respectively on the first sampling occasion. After this first sampling event, no Simuliidae were found at the SD1 site for the ensuing 3 week period at least (covering sampling events 2 and 3). Simuliidae were negligible at the SUM site, only occurring as one or two individuals (on average) at two out of the six sampling events, and we can therefore assume that this may have been the case at the first sampling event, had this site been sampled. Simuliidae slowly colonised at the STW site over the course of the wet season.

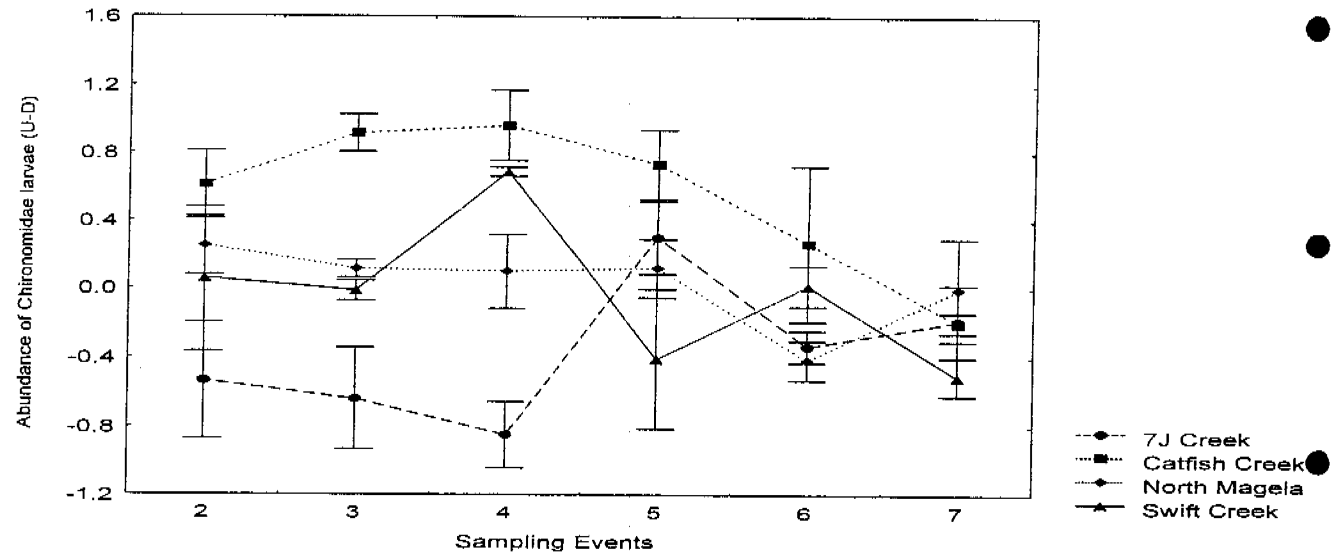


Figure 3.13 Chironomidae abundace difference data (upstream-downstream sites) for all streams over time (log10 transformed). Two-factor ANOVA showed F(15,48)=2.94;p<.002

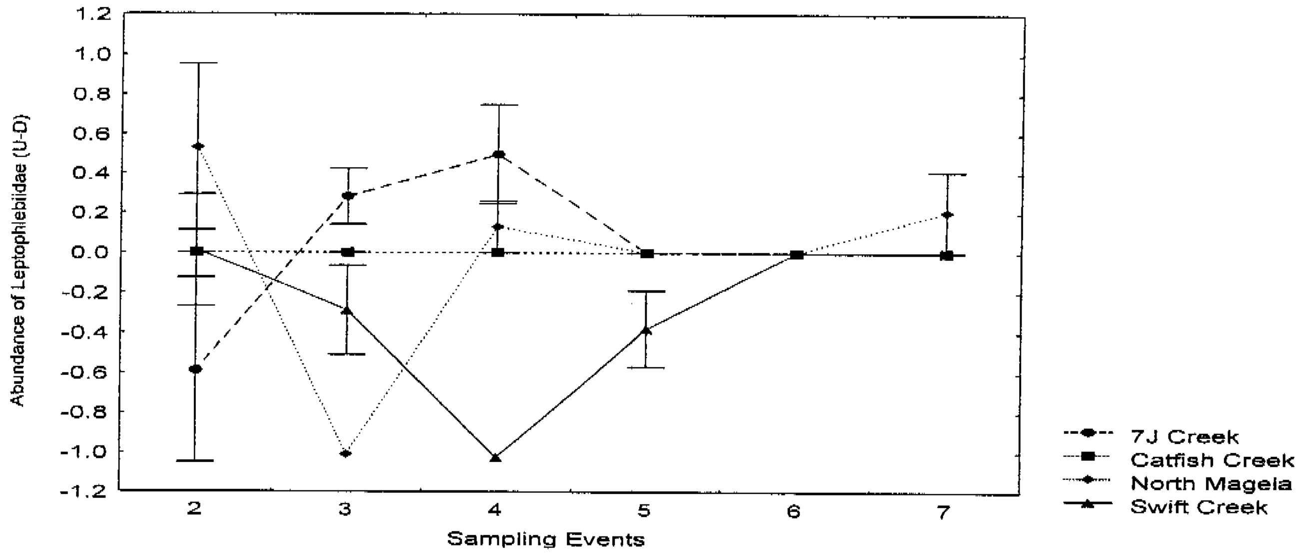


Figure 3.14 Leptophlebiidae abundance difference data (upstream-downstream) for all streams over time (log10 transformed). Two-factor ANOVA showed F(15,48) =4.85; p<.000

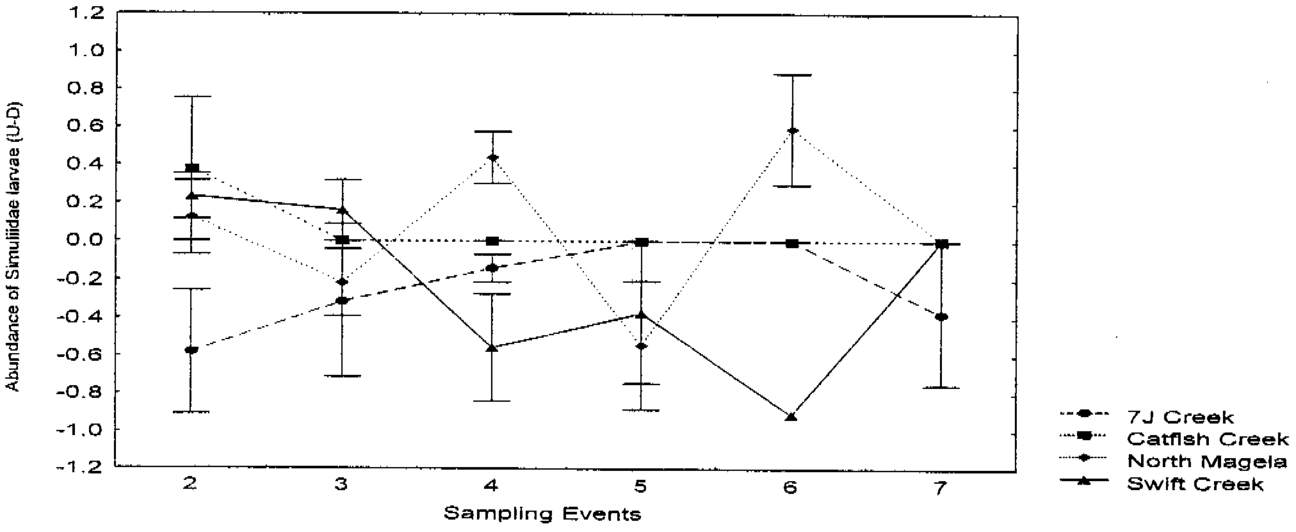


Figure 3.15 Simuliidae abundance difference data (upstream-downstream) for all sites over time (log10 transformed). Two-factor ANOVA showed F(15,48) =2.69; p<.005.

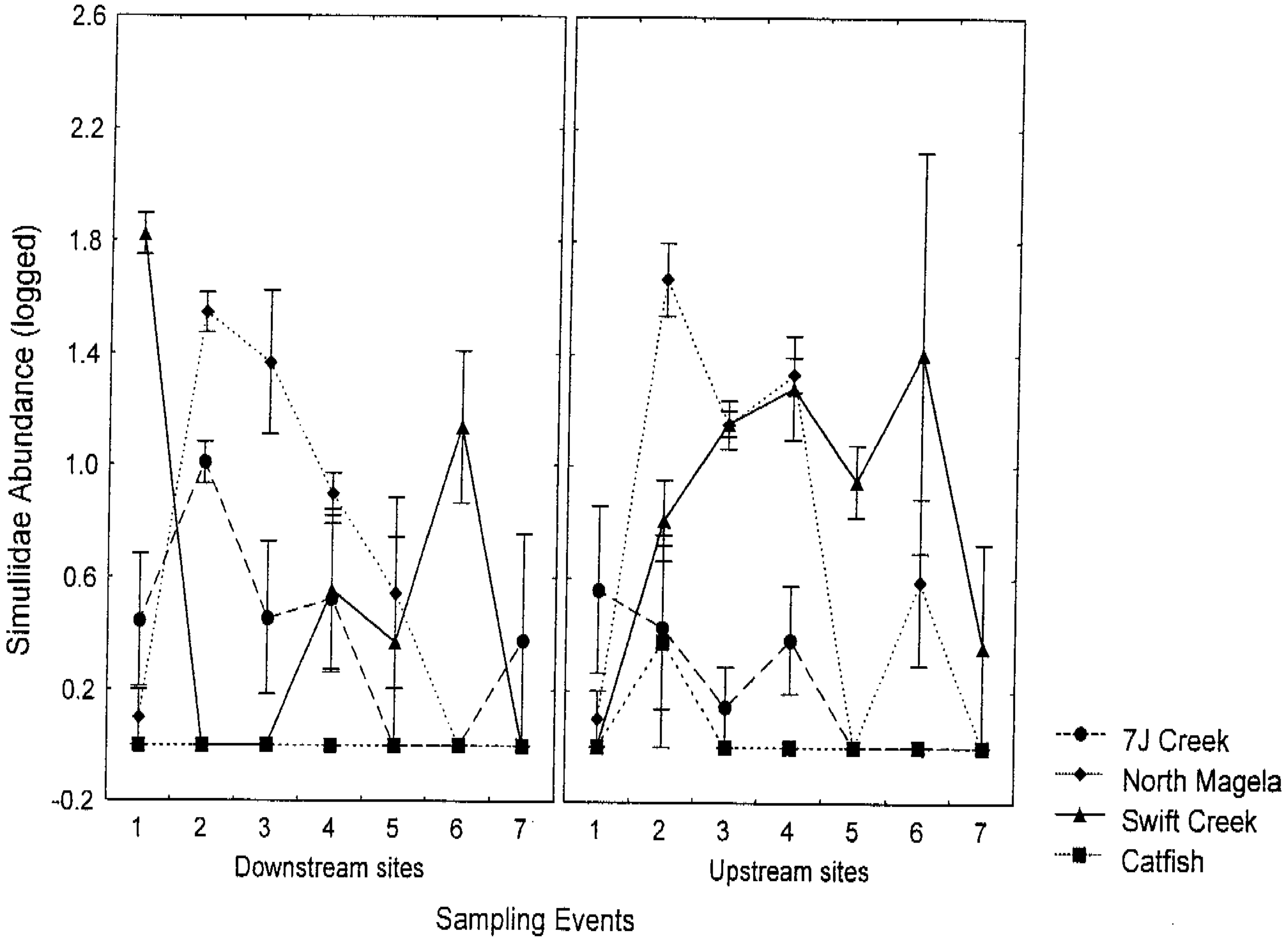


Figure 3.16 Simuliidae abundance at all upstream and downstream sites over time. Data has been $\log_{10}(x+1)$ transformed. STW is included as the Swift Creek upstream site in order to show the first sampling event. Three-factor ANOVA showed F (18, 112)=4.38; $p<.000$

Table 3.4 Values from ANOVA for all measures of paired-site difference

Variable	Effect	df Effect	MS Effect	df Error	MS Error	F	p-level
Dissimilarity	Creek	3	0.01	48	0.004	1.487	0.230
	Time	5	0.01	15	0.007	0.960	0.472
	Creek/Time	15	0.01	48	0.004	2.050	0.031
Abundance	Creek	3	2.07	48	0.125	16.568	0.000
	Time	5	0.42	15	0.336	1.262	0.330
	Creek/Time	15	0.34	48	0.125	2.684	0.005
Richness	Creek	3	4.53	48	14.667	0.309	0.819
	Time	5	25.38	15	31.899	0.796	0.569
	Creek/Time	15	31.90	48	14.667	2.175	0.021
Chironomidae	Creek	3	2.61	48	0.159	16.448	0.000
	Time	5	0.38	15	0.466	0.809	0.561
	Creek/Time	15	0.47	48	0.159	2.938	0.002
Simuliidae	Creek	3	0.55	48	0.157	3.475	0.023
	Time	5	0.09	15	0.423	0.206	0.955
	Creek/Time	15	0.42	48	0.157	2.687	0.005
Leptophlebiidae	Creek	3	0.37	48	0.107	3.452	0.024
	Time	5	0.14	15	0.520	0.269	0.923
	Creek/Time	15	0.52	48	0.107	4.853	0.000
Simuliidae	Creek	3	4.29	112	0.110	38.873	0.000
	U/D stream	1	0.13	3	0.405	0.315	0.614
	Time	6	1.18	18	0.858	1.371	0.279
	Creek/U-D	3	0.41	112	0.110	3.672	0.014
	Creek/Time	18	0.86	112	0.110	7.774	0.000
	U-D/Time	6	0.33	18	0.483	0.684	0.665
	Creek/U-D/Time	18	0.48	112	0.110	4.379	0.000

While many interactions were produced with the Tukey HSD test, the only ones of interest were between the upstream and downstream sites in each creek, on each of the separate sampling occasions. The only significant interactions were between Swift Creek on the first ($p<0.0001$) and third ($p<0.04$) sampling events, indicating that some factor, or factors other than chance, were responsible for producing such high numbers of Simuliidae at the SD1 site in relation to the STW site. There were no other significant interactions between upstream and downstream sites in the other creeks at any of the sampling events, over the sampling period. It is of interest that the SD2 site also recorded a marked decline in abundance after the first sampling event. The three replicates at this site in December found 770, 35 and 94 taxa, declining to very few individuals (1, 4 and 30) by the second sampling event.

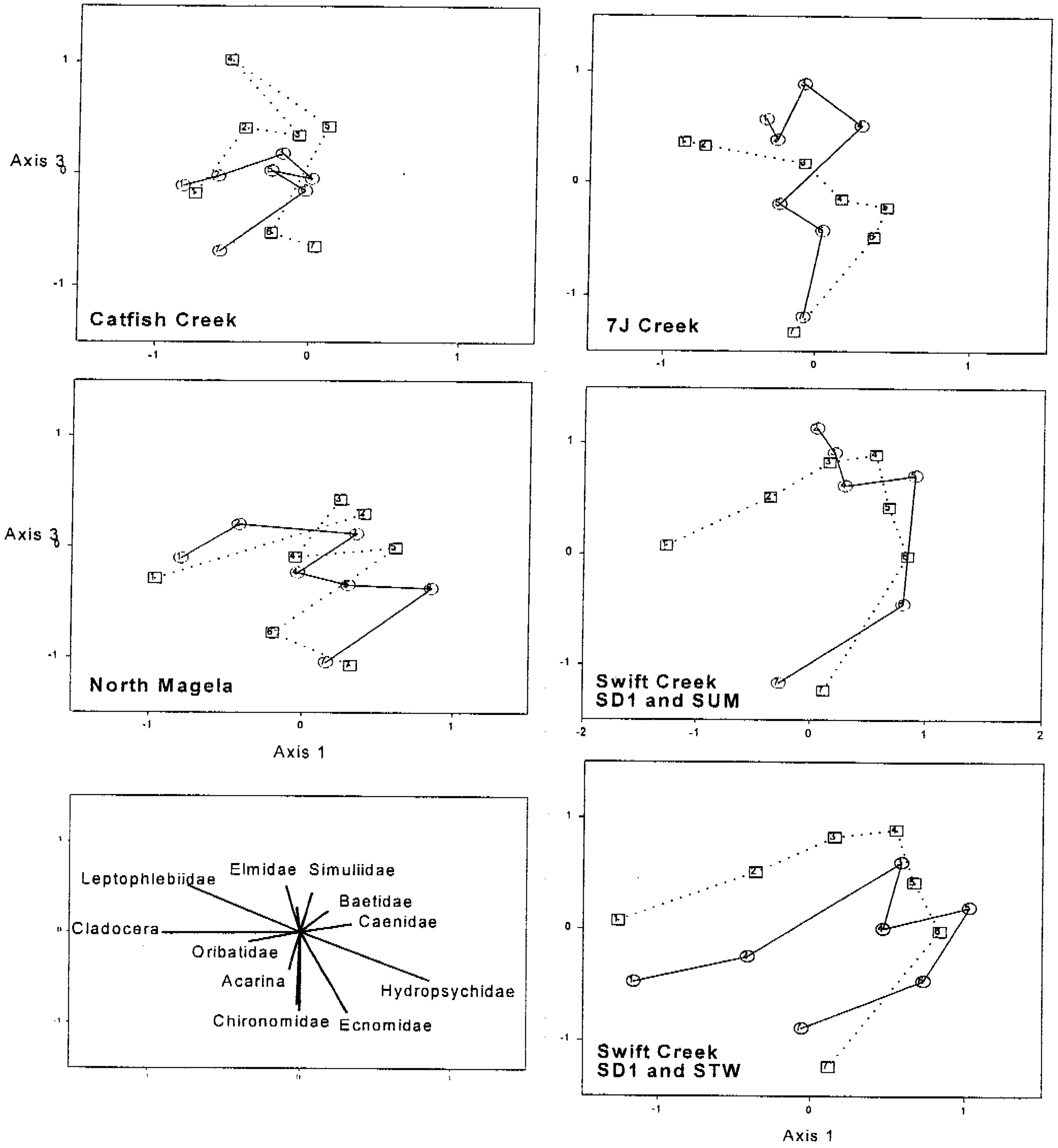
3.3.4 Multivariate ordinations showing seasonal patterns of all streams over the wet season

A single numerical output of multivariate ordinations, the stress value, indicates the validity of the ordination and is a measure of how "difficult" it was to place the dissimilarity points onto a reduced space. Stress values less than 0.2 are the preferred outcome and indicate that >80% of the relationship in the dissimilarity matrix has been explained in the ordination (Belbin 1993; Clarke 1993). Sometimes more than two dimensions may be required in order to reduce the stress level to an acceptable limit (Kenkel and Orloci 1986). While choice of dimensions is a fairly subjective process, it is recommended that the number of dimensions be increased until stress <0.2 is reached, as this produces the highest correlation with the environmental variables, ensuring important relationships are not missed (Faith and Norris 1989). Three dimensions were required for the main ordination described below, in order to reduce the stress level below 0.2.

3.3.4.1 Seasonal trends in taxa succession

Results from the multivariate ordination, based on all sampling sites at all sampling events over the wet season, produced a picture of seasonal trends for the different creeks. The axes which best represent the trends are axis 1 vs axis 3 for seasonal community structure and habitat characteristics, and axis 1 vs axis 2 for environmental parameters. It is worth noting that the vectors produced (i.e. correlates of ordination space) are projecting into 3 dimensional space upon a 2 dimensional diagram, thereby giving an effect of differing lengths. This is an artifact of the situation and does not relate to their differing influence upon the ordination (Faith and Norris 1989). It is also worth noting that this makes their interpretation slightly more difficult than for 2 dimensional space.

Changes in community structure for each of the creeks over the sampling period are presented in Figure 3.17. Even though the ordination is based on all data, the different creeks are shown as separate plots for ease of interpretation. All the creeks displayed a



Figures 3.17 SSH-MDS Multivariate ordination in 3 dimensions of all sampling sites over the wet season based on $\log_{10}(x+1)$transformed data. (Stress = 0.18). Macroinvertebrate taxa significantly correlated ($p<0.01$) are shown. Squares indicate downstream and circles upstream sites.

pattern of “arch-shaped” trajectories with varying degrees of definition. Catfish Creek and both Swift Creek combinations (SD1 vs SUM and SD1 vs STW) produced the most precise examples showing a solid inner circle of points (U/S) surrounded on the outside by a dotted circle of points (D/S). This indicated a temporal succession of taxa initiated at the upstream site and "followed" by the downstream site. The value produced for the fourth sampling event at Catfish Creek downstream shows the community disruption that occurred following a severe spate. (This value was included in most of the analysis, except for ANOVA, where it was considered an outlier as it sat outside a 95% confidence interval with the other values). 7J displays good pattern definition for the downstream site, while North Magela produced the least evident trend.

Catfish Creek demonstrated a much tighter fit around the arch than the other Creeks and also closer biological spacing between the 1st and 7th sampling events. Catfish is spring fed throughout part of the dry season and this tighter fit of the arch may be indicating the smaller "dry" period it experiences in relation to the other creeks.

3.3.4.2 Taxa significantly correlated with the ordination space

Taxa significantly correlated with the ordination space (all $p<0.01$) are also shown in Figure 3.17 and their vectors produced a picture of seasonal change. Amongst the significant early colonisers were the Cladocera and Oribatid mites, seen to dominate at North Magela, Catfish Creek and Swift Creek (SD1 vs SUM). Leptophlebiidae were another significant taxon early on, and, together with Elmidae and Simuliidae ($p<0.03$) all appeared to benefit from the high flow conditions. These taxa influenced the ordination when water levels and stream velocity were variable, yet generally high in all streams, corresponding to sampling occasions two and three.

The Baetidae and Caenidae mayflies were fairly dominant as a core taxa group throughout the wet season, influencing community structure particularly around the fourth and fifth sampling periods. The caddisfly families, Hydropsychidae and Ecnomidae showed most influence on the ordination around the sixth sampling occasion in mid April when macrophytes, the preferred substrate for these filter feeders to attach, had reached maximum growth and diversity.

The ordinations show that, for all the creeks, Chironomidae dominated the samples by the seventh sampling event. These taxa are also often present in early recolonisation patterns (Gunn 1997; Paltridge 1992) and their influence on the ordination here shows a return to pre-flow conditions. Acarina were also influential in the recessional flow period.

The arch-shaped trajectories displayed by all the Creeks indicate that the most influential taxa are "pulling" the biological picture around as succession occurs over the wet season.

3.3.4.3 Environmental and Habitat Variables

Figure 3.18 shows the same ordination as described above (axis 1 vs 3) with the significant environmental vectors of water depth, alkalinity and pH imposed on the diagram.

Axis 3 appears to depict a gradient of water velocity/discharge and hence this axis separates still vs flowing water fauna. Water depth, alkalinity and pH are surrogate measures which correlate with water velocity and discharge, and influence the ordination in opposing directions along axis 3. High velocity correlates positively with high water depth. Water flow correlates negatively with alkalinity and a higher (more neutral) pH, found to be influential towards the end of sampling during recessional flows. As water flow is reduced towards the end of the wet season, solutes concentrate and water quality moves away from the typically acidic conditions of rain water (Humphrey et al 1990). Paltridge et al (1997) also found water velocity produced a significant axis of site separation between lentic and lotic habitats in Magela Creek.

The nature of the successional "loop" observed in the creeks in the Axis 1 vs Axis 3 plot (Fig 3.18) implies that these two axis are intimately linked. Axis 1 most strongly reflects faunal succession, as well as hydrological regime of the creeks. Sites progress through time across this axis, reach a successional "peak", then loop back to a fauna that increasingly resembles early recolonisation. Although "time" is correlated with the ordination space along this axis, this correlation is negated to some extent by the return of late wet season samples to an early wet season appearance. (In the PCC component of PATN ordination, correlations are constrained to linear models).

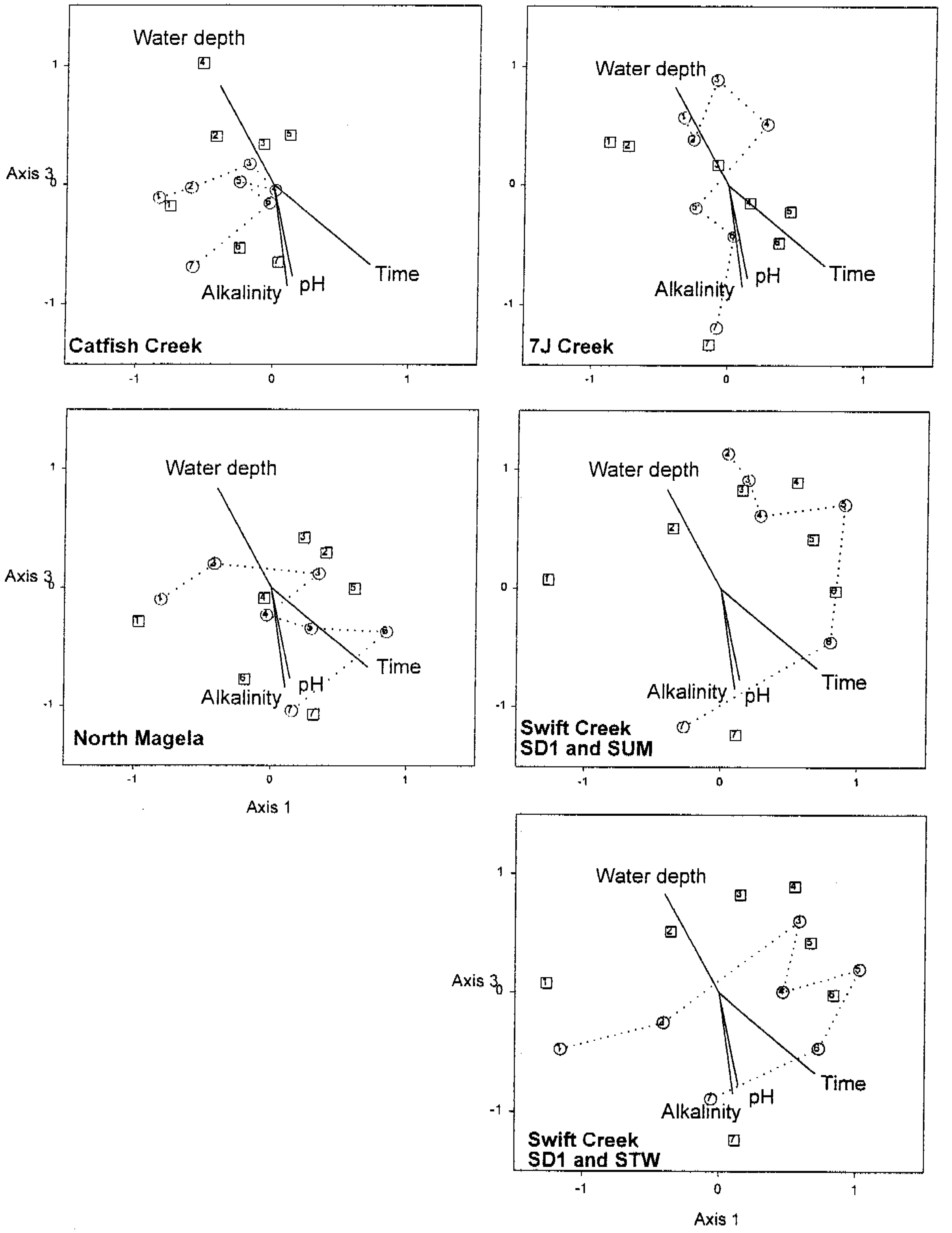


Figure 3.18 SSH-MDS Multivariate ordination in 3 dimensions of all sampling sites over the wet season based on $\log_{10}(x+1)$ transformed data (stress=0.18). Habitat variables significantly correlated ($p<0.01$) are shown. Squares indicate downstream and circles upstream sites, and the upstream sites are connected by a dotted line.

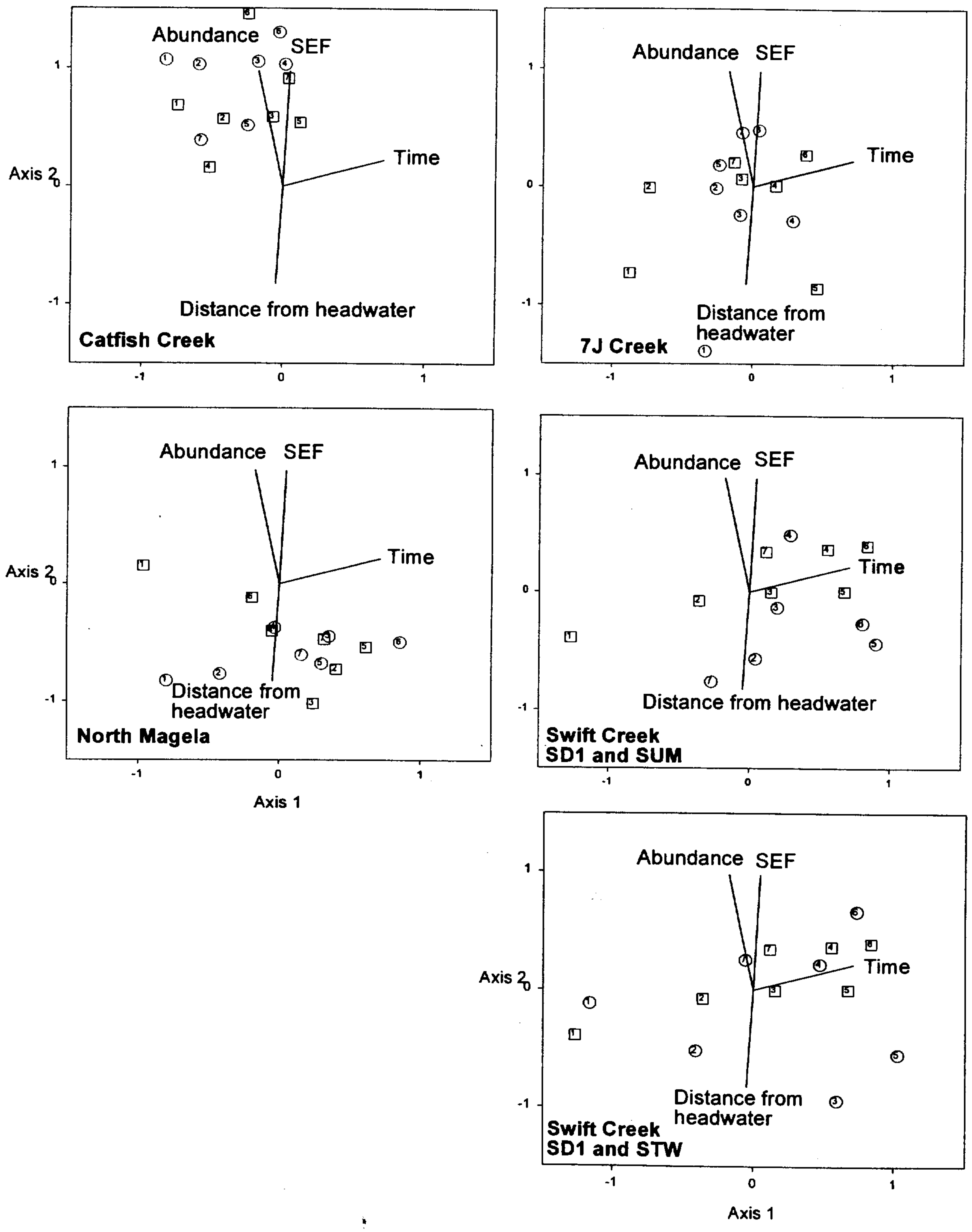


Figure 3.19 SSH-MDS Multivariate ordination in 3 dimensions of all sampling sites over the wet season based on $\log_{10}(x+1)$ transformed data (Stress=0.18). Environmental variables significantly correlated ($p<0.01$) are shown. Squares indicate downstream and circles upstream.

As with axis 3, this axis also appears to reflect a still/slow water (to the left, early and late flow) vs lotic/fast water (to the right, main flow period) gradient in macroinvertebrate composition. The extent of the trajectory loop across the axis also appears to depend upon the degree to which the creeks are truly seasonal in nature; the more seasonal the stream, the greater the compositional changes observed over the wet season. Thus, samples from Catfish Creek, a system that flows for much of the year, lie to the left of the ordination, indicating a faunal composition that does not vary as greatly throughout the wet season compared with the other creeks.

The same ordination is presented in Figure 3.19 this time with axis 1 and 2 displayed. Significant correlates of ordination space are; total macroinvertebrate abundance, SEF (Submerged and Emergent Feathery macrophytes; *Eriocaulon* and *Eleocharis caespitosissima*, both well established at the upstream site at Catfish Creek before sampling commenced), and distance from headwaters. Site separation along axis 2 therefore directly relates to the distance from the recolonising source of much of the taxa (located at the headwaters of each stream), which, in turn, influences abundance at each site.

Catfish Creek sites were located nearest to the spring-fed headwaters of the creek, and consequently showed significantly higher macroinvertebrate abundance at both sites, especially the upstream site. A further correlation with this axis was macrophyte density; this was highest in Catfish Creek where large, and established, stands of macrophytes occurred early in the wet season. In contrast to this situation and, at the other end of the axis 2 gradient, North Magela showed lowest macroinvertebrate abundance at both sites overall, and was the stream located furthest from the recolonising source. Swift Creek and 7J are situated at approximately the same distance from their headwaters and showed similar macroinvertebrate abundance over the wet season.

3.3.4.4 Seasonal trends in taxa succession - sampling events separated

Ordinations performed on samples from all creeks, but separately for each sampling occasion, showed the actual biological spacing amongst the streams in greater detail (Fig 3.20). At the first sampling event in late December, the Swift Creek downstream sites (D1 and D2) were separated from the other sites by the influence of Simuliidae larvae.

There was a stronger gradient along axis 2, where the early colonising Cladocera and water mites (Acarina and Oribatidae) separated Catfish Creek from all the other sites. Catfish Creek also showed the closest biological spacing between upstream and downstream locations. Early establishment of flow in Catfish Creek would have carried recolonising taxa downstream, establishing similar macroinvertebrate community structure at the downstream site prior to the other streams. 7J and North Magela creeks revealed a similar ecological distance in ordination space between upstream and downstream locations, while Swift Creek (SD1 vs STW) sites were located furthest apart. The SUM site was not sampled at this time.

The SUM site was sampled for the first time on the second sampling occasion in January and it sat in a completely different biological space to the other Swift upstream site (STW). Catfish Creek sites clearly occupied a distinct ecological space to the other streams for sampling events two and three.

The fourth sampling event displayed a spread of influence, as opposed to the previous sampling occasions where Catfish Creek sites strongly influenced the ordinations. The downstream site at Catfish Creek is evidently different, and correlates in a negative direction with flow conditions. Leptophlebiidae was an important taxon for the two Swift downstream sites, separating them out from the main group. The Swift upstream main (SUM) site was largely influenced by root mats during this sampling occasion and it is plausible that the heavy rains, which occurred just prior to this sampling, exposed the banks to some erosion.

The fifth sampling event showed Chironomidae producing a gradient along axis 1, with the two downstream sites in Swift Creek located more towards the negative end of this gradient. The sixth sampling event again shows the dominant taxa influencing Catfish Creek to a greater degree than the other streams. Catfish Creek was predominantly influenced by Cladocera and Corixidae by the seventh sampling event in early May, and Chironomidae exerted an influence on all sites except for North Magela and SUM.

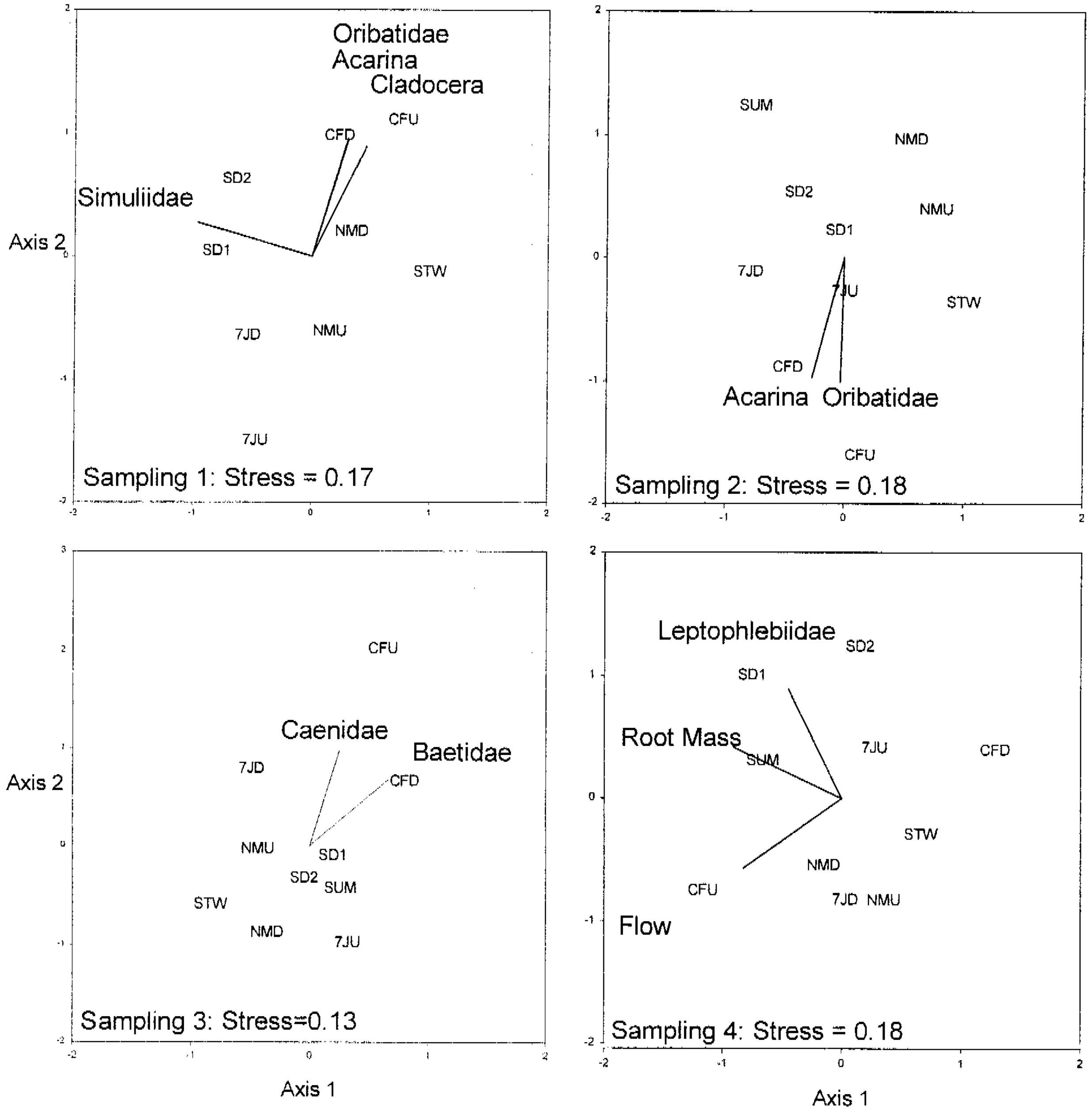


Figure 3.20 SSH-MDS Multivariate ordinations in 2 dimensions of all sampling sites over the wet season on separate sampling occasions 1 - 7. Data has been $\log_{10}(x+1)$ transformed. Macroinvertebrate taxa, habitat and environmental variables significantly correlated ($p<0.01$) are shown. Sites are indicated according to the codes described in section 2.1.4

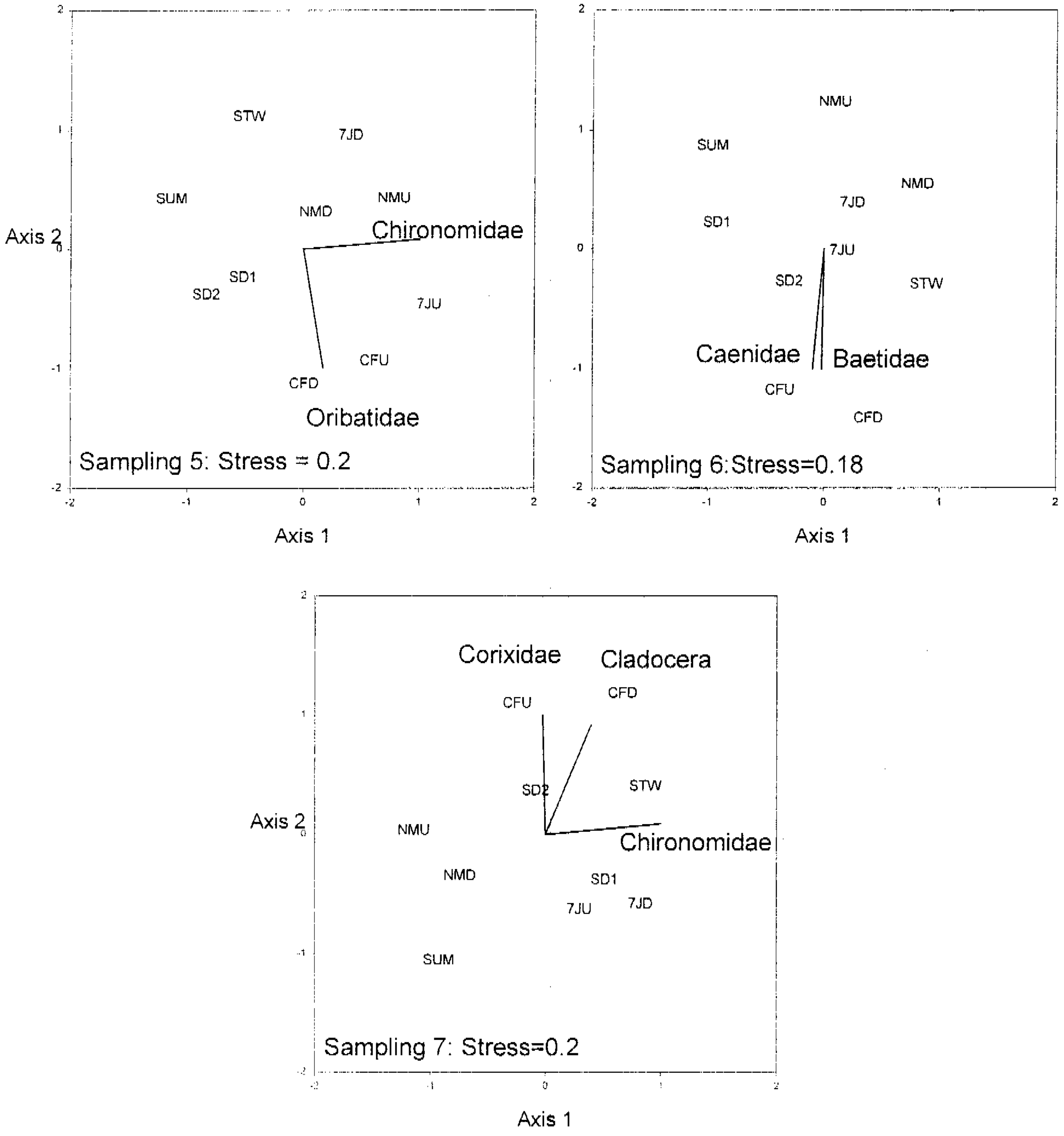


Figure 3.20 continued. Sampling events 5-7 shown

The large ecological difference between both Swift downstream sites and SUM is noticeable.

3.3.4.5 Summary of separate ordinations

The trends displayed at each sampling event provide a clear picture of the biological spacing between upstream and downstream sites at each creek over the wet season (Fig 3.20). At the beginning of the wet season, Swift D1 and D2 were separated out from the other sites, by Simuliidae. They were again separated out at the fourth sampling occasion by Leptophlebiidae and, to a lesser degree, with a negative correlation to Chironomidae on the fifth occasion. Apart from these observations, the two Swift downstream sites did not sit outside of the range presented by the control streams. The two upstream sites at Swift Creek showed markedly different patterns over the wet season, with the SUM site showing most similarity to the downstream sites, with the exception of the seventh sampling event.

North Magela represented a good control creek due to the close relationship maintained between upstream and downstream sites throughout the sampling period. Catfish also maintained a tight relationship (apart from the downstream site at the fourth sampling event) but clearly occupied a distinct biological space to the other sites.

3.3.5 Ecological trends over the wet season

3.3.5.1 Seasonal changes in the major macroinvertebrate taxa

The contribution of the different taxa groups to seasonal changes in macroinvertebrate communities is shown by way of histograms in Figures 3.21-3.24. These figures corroborate the results shown in the ordinations (described above). In the early recolonisation, most of the sites were dominated by microcrustaceans and "Other Taxa" (refer to Table 3.5). The microcrustaceans contributed less to total abundance after the first and second sampling events for the creeks, indicating that they are early recolonisers. The worms, Oligochaetes and Nematodes, dominate in the Other Taxa group for Swift Creek (see Appendix 8) and the other control streams (7J and North Magela), whereas water mites, Acarina and Oribatidae, dominate for Catfish Creek. Of

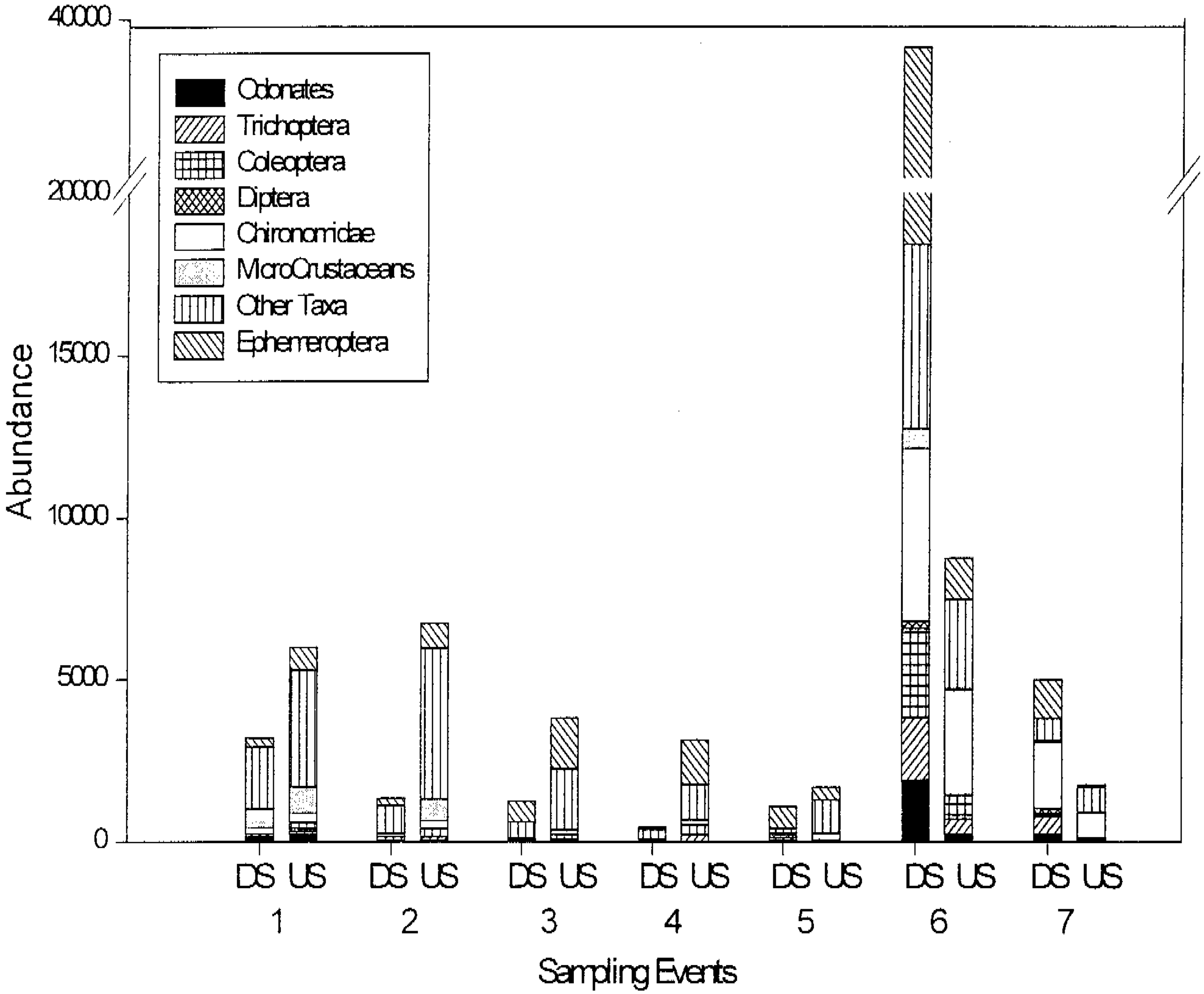


Figure 3.21 Taxa families grouped into Orders from both sites in Catfish Creek over the wet season.

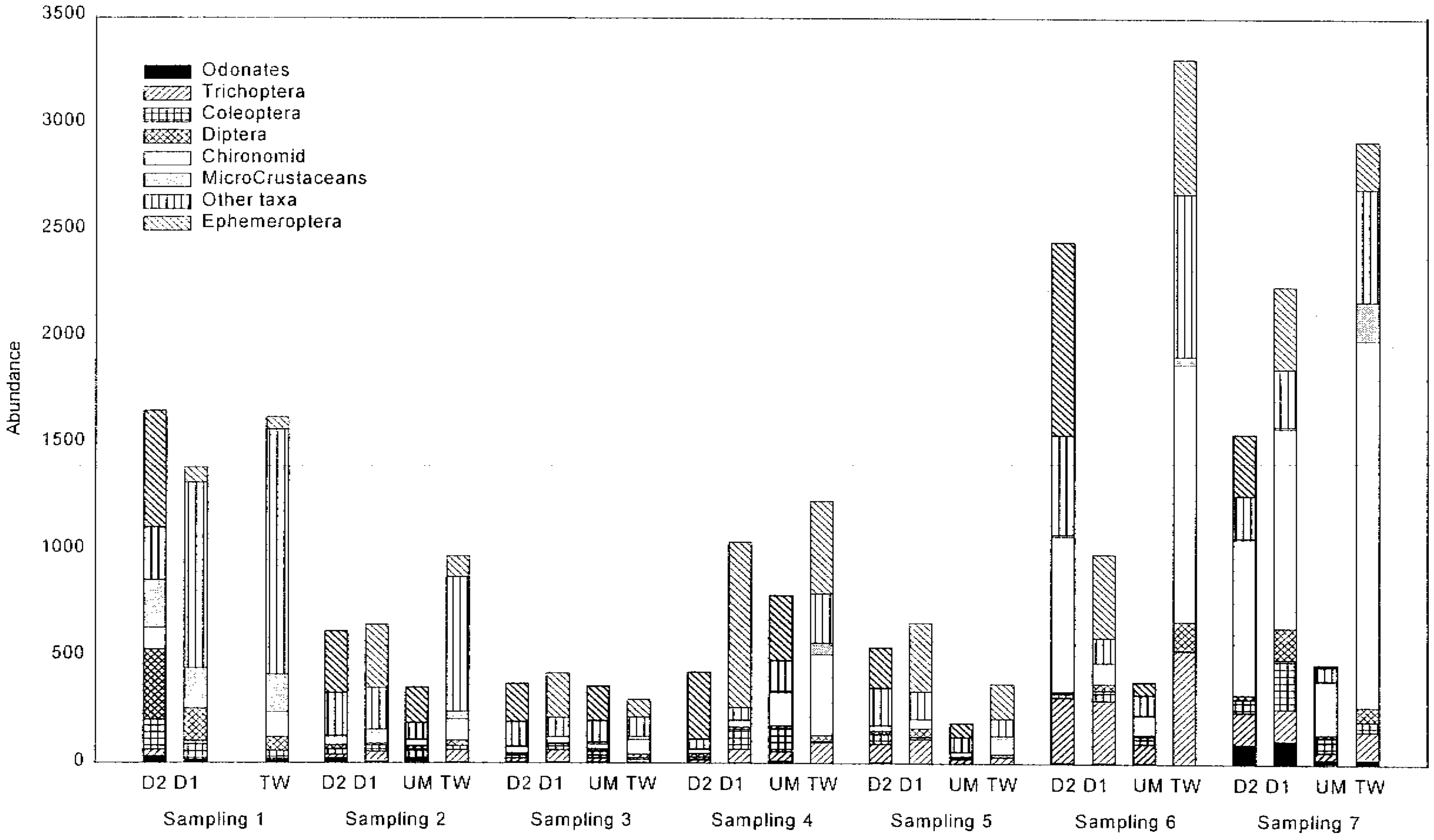


Figure 3.22 Taxa families grouped into Orders from all sites in Swift Creek over the sampling period

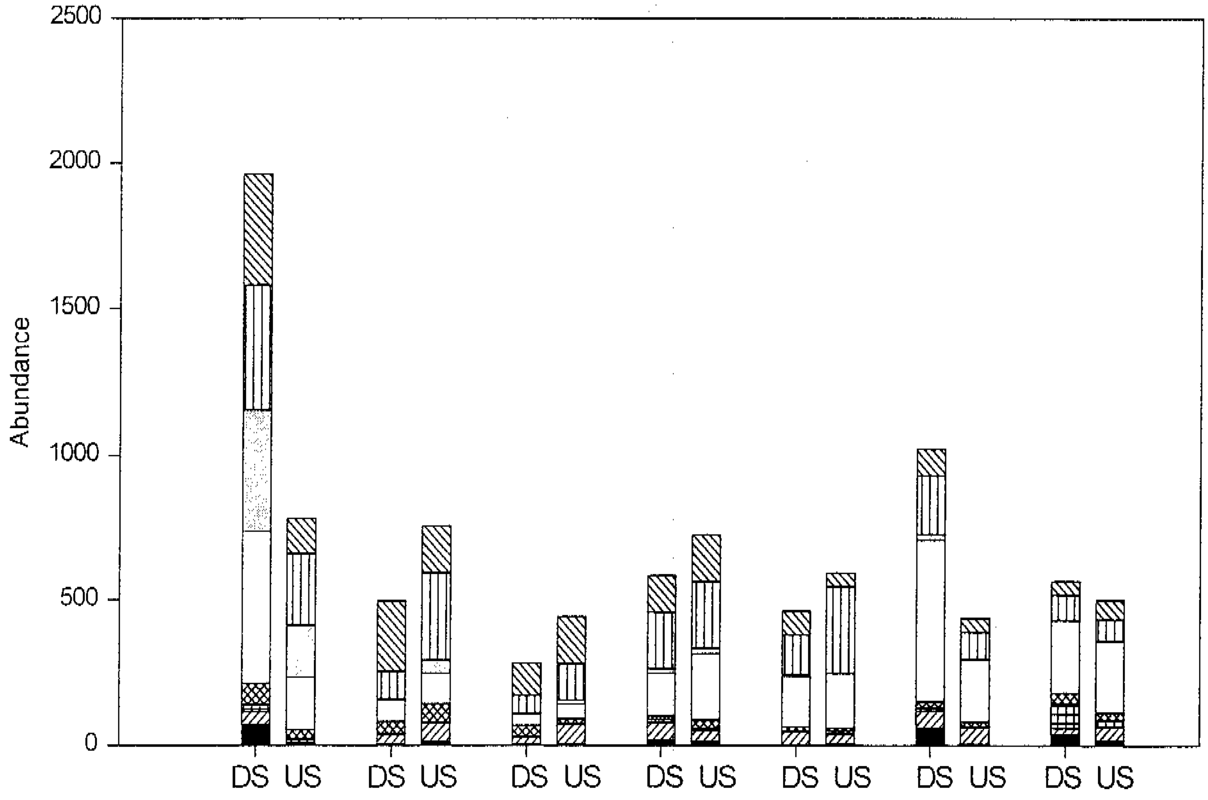


Figure 3.23 Taxa families grouped into Orders from both sites in North Magela over time

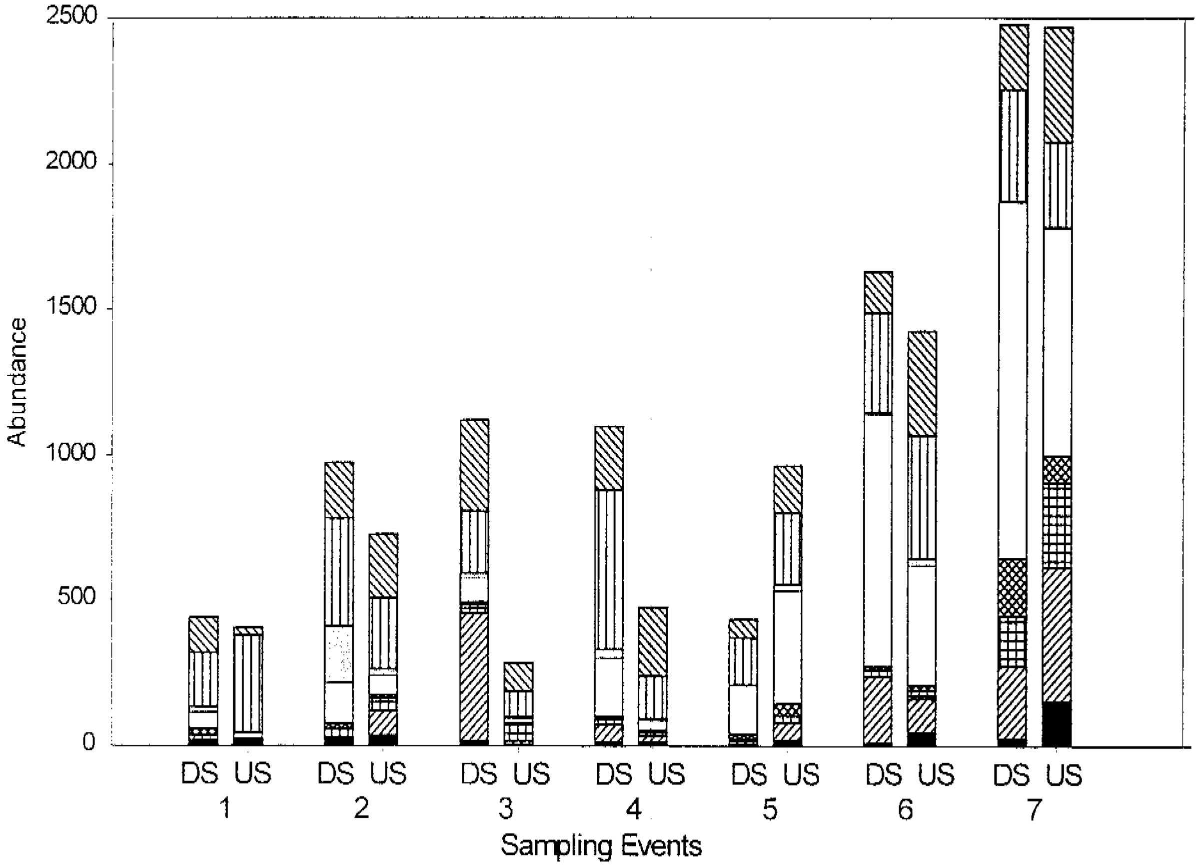


Figure 3.24 Taxa families grouped as orders from both sites in 7J Creek over time

interest is that North Magela downstream (Fig 3.23) and Swift downstream 2 (Fig 3.22) both have high contributions from Ephemeroptera (mayflies) as well, and, as revealed by the graphs, these taxa remain consistently part of the dominant taxa over the sampling period for all of the streams.

Table 3.5 Taxa families grouped into Orders (except where noted)

Taxa Order	Taxa families
***Odonata**	Anisoptera, Coenagrionidae, Corduliidae, Gomphidae, Isosticidae, Libellulidae, Protoneuridae, Zygoptera
Trichoptera	Ecnomidae, Hydropsychidae, Hydroptilidae, Leptoceridae
Coleoptera	Dytiscidae, Elmidae, Hydraenidae, Hydrophilidae, Noteridae, Scirtidae, Staphylinidae
Diptera	Ceratopogonidae, Culicidae, Simuliidae, Tabanidae, Tipulidae
***Chironomidae**	Chironomidae larvae and pupae
***Microcrustacean**	Cladocera, Copepod, Ostracod
***Other taxa**	Acarina, Amphisipododidae, Ancylidae, Atyidae, Belostomatidae, Collembola, Corixidae, Gerridae, Hydrometridae, Mesovellidae, Naucoridae, Nematodae, Nepidae, Notonectidae, Oligochaete, Oribatidae, Palaeomonidae, Pleidae, Pyralidae, Sundathelphusidae, Veliidae
Ephemeroptera	Baetidae, Caenidae, Leptophlebiidae

* Odonata includes Anisoptera and Zygoptera (both sub-orders).

* Chironomidae (Diptera) is the family level

* Microcrustacean includes Cladocera (sub-order), Ostracod (sub-class) and Copepod (sub-class).

* Other taxa includes: Acarina (Order), Oribatidae (sub-order), Oligochaete (class), Nematode (order), and Collembola (order). (Williams 1980).

Most notable for Swift Creek and 7J Creek, Trichoptera (caddisflies) occurred at the second sampling event and remained fairly stable until the sixth where they increased in number (Fig 3.24). Odonates (dragonflies) do not contribute greatly to the overall taxa abundance, but can be seen to increase their contribution to community abundance towards the end of the wet season at all sites. They were also found during the first sampling event for Swift Creek, Catfish Creek and North Magela at the downstream site. These taxa, together with several of the Hemipterans (True bugs, Noteridae and Belostomatidae) are voracious predators and find prey more concentrated by the recessional flow period.

Coleopterans (beetles) reflect the above trend described for dragonflies, dominating early on in the sampling regime, and then again towards the end, indicating their preference for lentic water. At Catfish Creek, these taxa were found in low numbers during the final sampling event (Fig 3.21), whereas for 7J and Swift D1, they formed a substantial part of the fauna. The largest influence on abundance by the end of the wet season is from the Chironomidae, which become established by the end of April. This pattern is reflected across all sites within all the creeks.

At Catfish Creek (Fig 3.21), it is obvious that there was a dominance of the Ephemeropterans, and Other Taxa (accounted for by Acarina and Oribatidae) until the final two samplings where Chironomidae also began to dominate. This trend was not reflected in any of the other creeks.

3.3.5.2 Taxa Richness

While the creeks sites all showed varied patterns of taxa richness (number) over time, they all produced taxa numbers that fell within the same range (Figs 3.25-3.28). In the early flow period, Swift Creek (Fig 3.26), North Magela (Fig 3.27) and Catfish Creek (Fig 3.25) all had reasonably high species richness. Swift Creek showed a noticeable decline in richness over the mid-wet season reaching lowest numbers at the fourth sampling event for all sites. An increase in richness commenced after this for all the Swift Ck sites, apart from the downstream 2 site. This site had low taxa richness until the final sampling event, by which time a dramatic increase was noted. By the recessional flow period, the two downstream sites had very similar numbers of taxa to the early sampling events.

North Magela showed a fluctuating pattern, also ending up with greater taxa richness by the recessional-flow period (Fig 3.27). Both upstream and downstream sites tracked each other closely for this creek. Apart from the first and last sampling occasions, the 7J sites also displayed a similar pattern of taxa richness over the wet season (Fig 3.28). The discrepancies in effect led to the upstream site showing a general increase in taxa richness over time, whereas the downstream site showed greater fluctuation with no real increase in richness by the recessional flow period. The trend at Catfish Creek was also one of less fluctuation in richness over the sampling period, apart from the fourth sampling event at the (flooded) downstream site where relatively fewer taxa were collected.

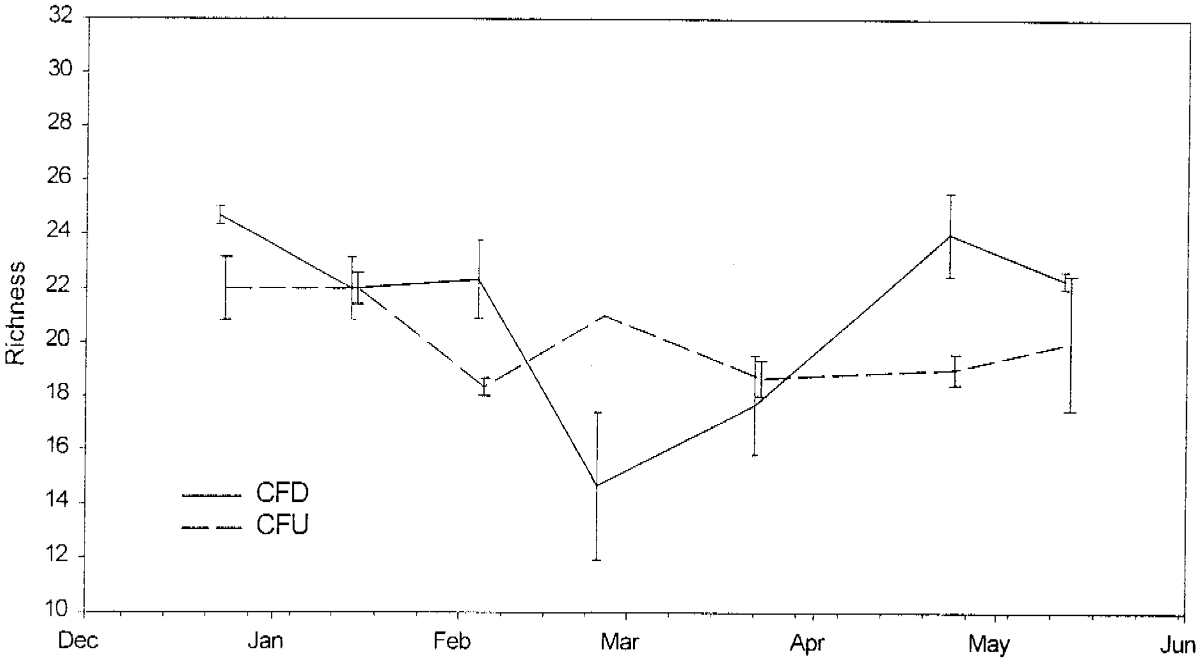


Figure 3.25 Taxa richness in both sites in Catfish Creek over the wet season (All graphs show standard error bars).

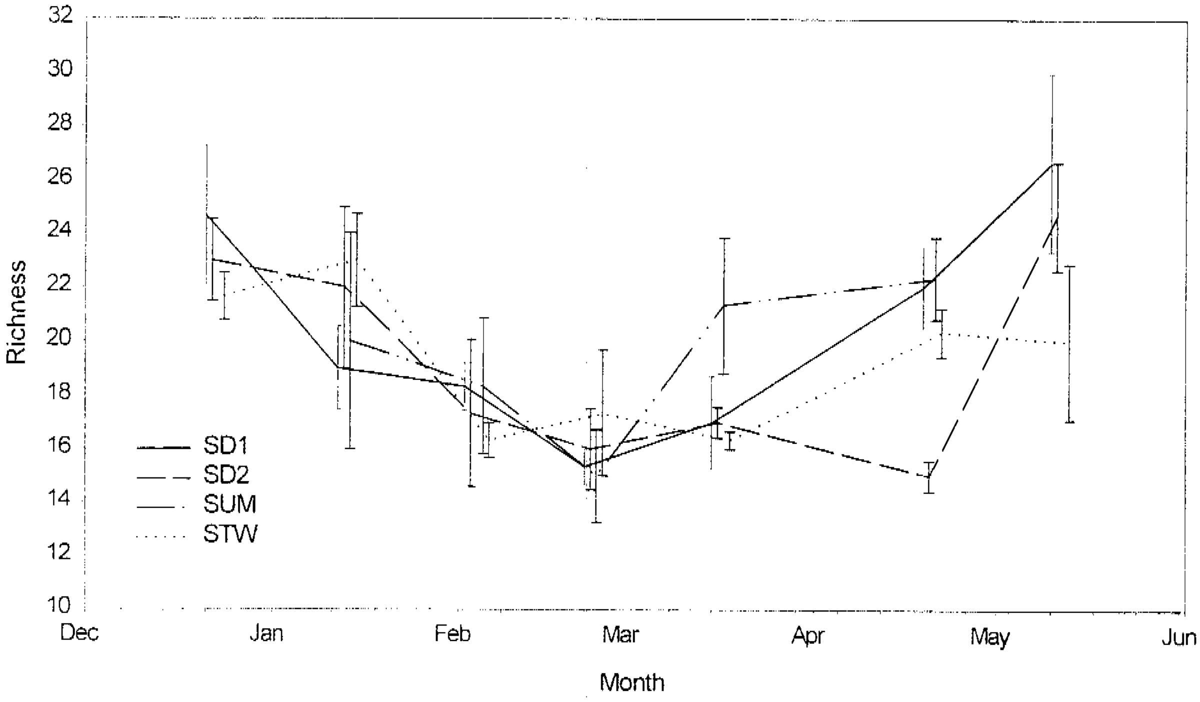


Figure 3.26 Taxa richness in all sites in Swift Creek over the sampling period.

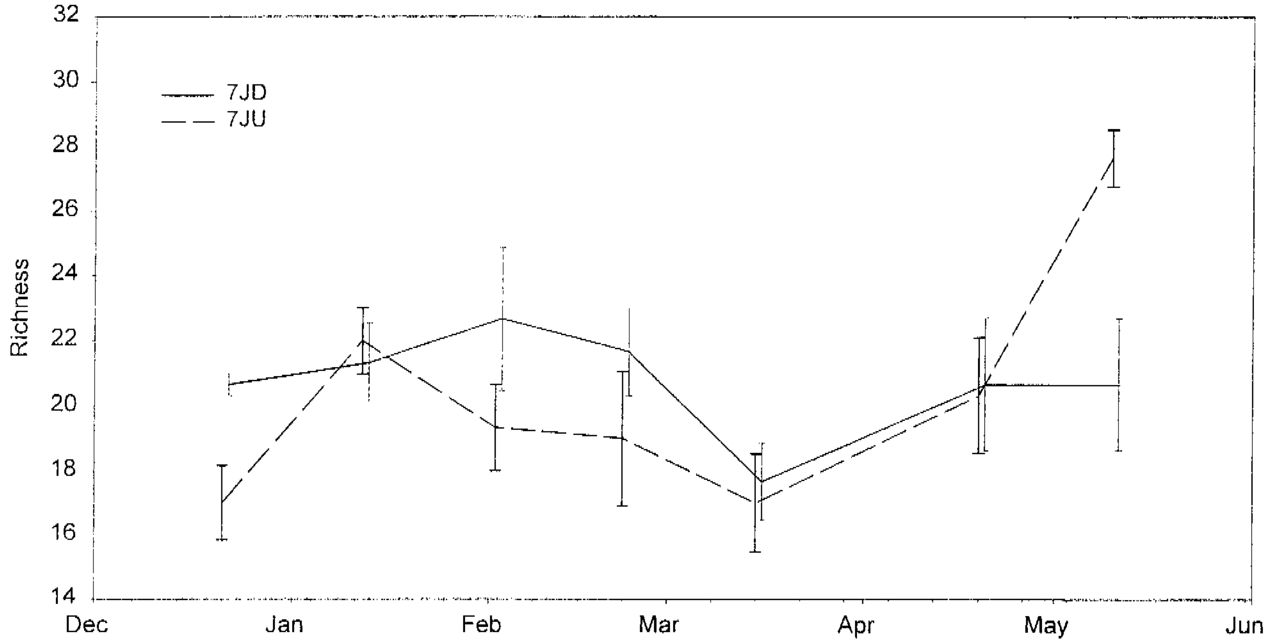


Figure 3.27 Taxa richness in both sites in 7J Creek over the wet season

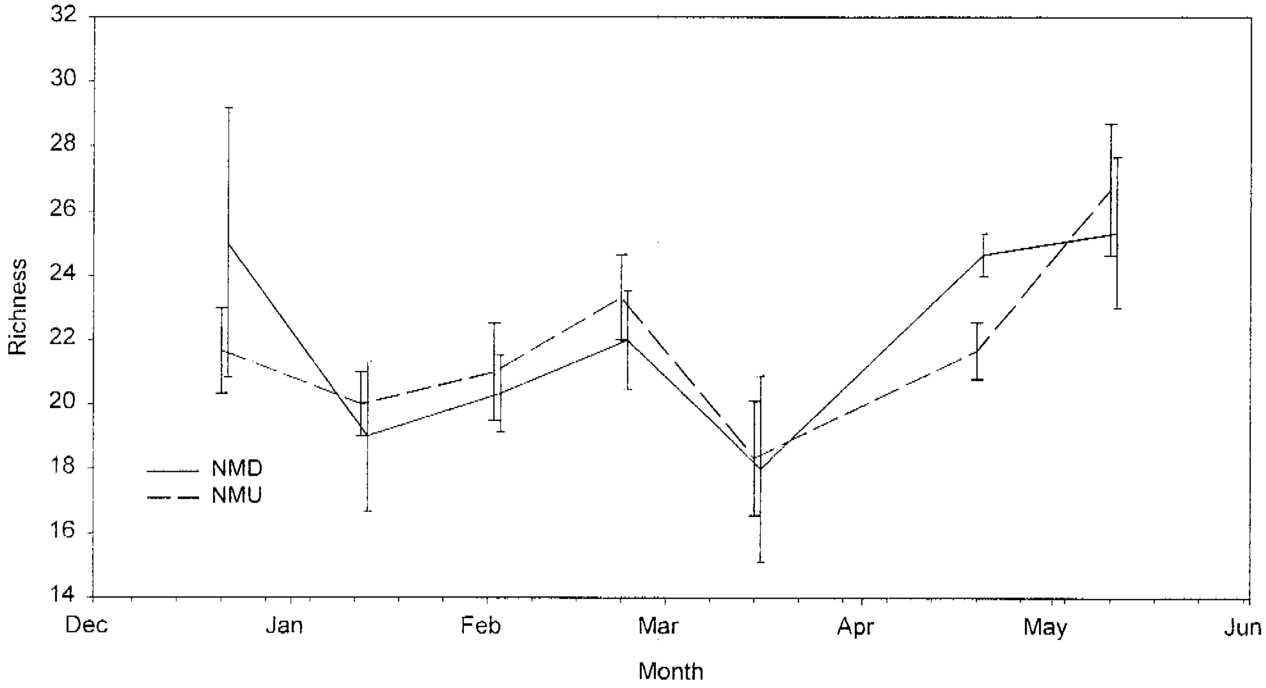


Figure 3.28 Taxa richness in both sites in North Magela over the wet season

3.3.5.3 Total Macroinvertebrate Abundance

7J was the only stream from this study which showed a general increase in macroinvertebrate abundance over the wet season, with both sites tracking one another quite closely (Fig 3.32). In the other creeks and with only a few exceptions, abundance patterns between upstream and downstream sites were also similar over the wet season. For Catfish and North Magela creek sites (Figs 3.29 and 3.31 respectively), however, abundances remained relatively unchanged over time while in Swift Creek, abundances were highest at the start and finish of the wet season (Fig 3.30). Overall, abundances were highest in Catfish Ck, then Swift, 7J and North Magela creeks.

3.3.5.4 Cumulative taxa richness

Fewer taxa were recorded for Catfish Creek compared to the other sites (Fig 3.33). The other three creeks showed similar patterns of increasing taxa accretion over the wet season. 7J Creek recorded 56 taxa from the downstream site and 50 taxa from the upstream site over the entire wet season. This was comparable to Swift Creek which contained 55 taxa at the D1 site, 51 at D2, 54 at SUM and 52 from STW. The highest number of taxa found, (57), were at North Magela Creek at both upstream and downstream sites. In contrast to this, Catfish Creek only recorded 45 taxa from the downstream site and 46 from the upstream location. For most of the wet season, taxa accretion was generally highest at the downstream sites for all creeks.

3.3.6 Ordination of the mine site tributaries and the sixth sampling event

An ordination was performed on the data combining sites from the four main creeks on the sixth sampling event with those from each of the tributaries draining the mine site into Swift Creek, sampled a few days later. This was undertaken to determine whether there was any gradient of disturbance that may have been present as a result of mine run-off. The water quality parameters for the tributaries can be found as Appendix 7.

The tributaries and main creeks produced an ordination with a stress level of 0.17 using three dimensions (Figs 3.34 and 3.35). The environmental parameters which correlated significantly with the ordination space were NotF (all plant types except *Eleocharis* sp

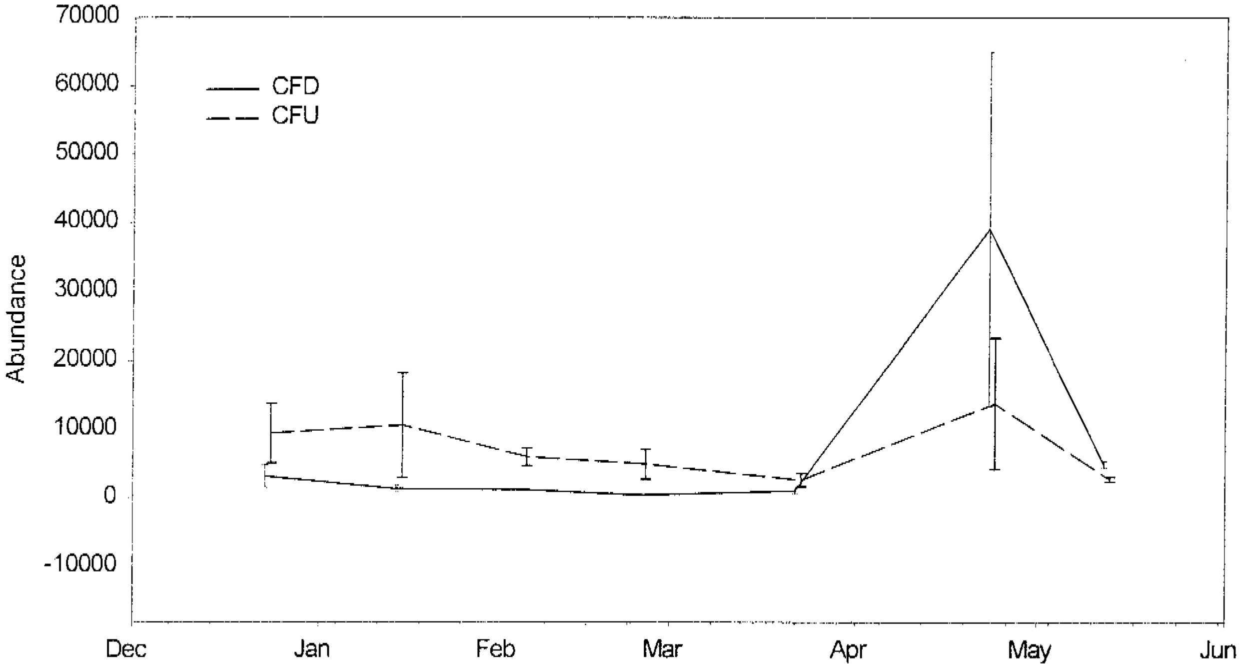


Figure 3.29 Taxa abundance at both sites in Catfish Creek over the wet season (All graphs show standard error).

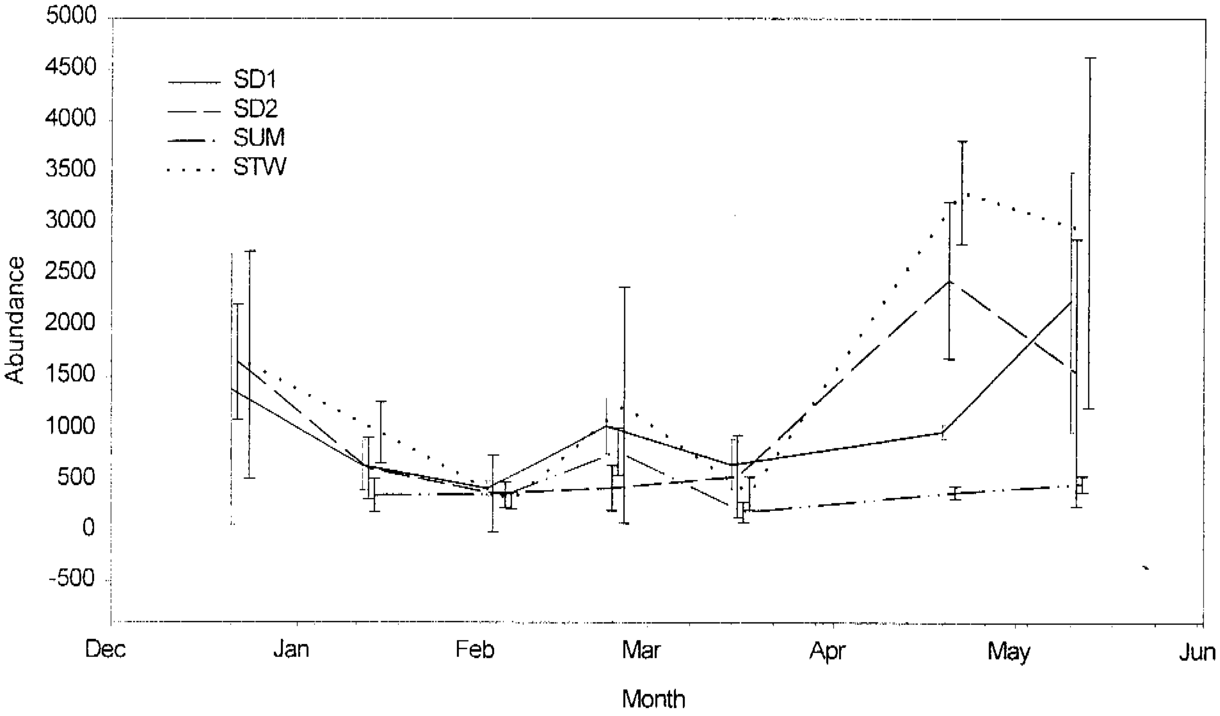


Figure 3.30 Taxa abundance at all sites in Swift Creek over the wet season.

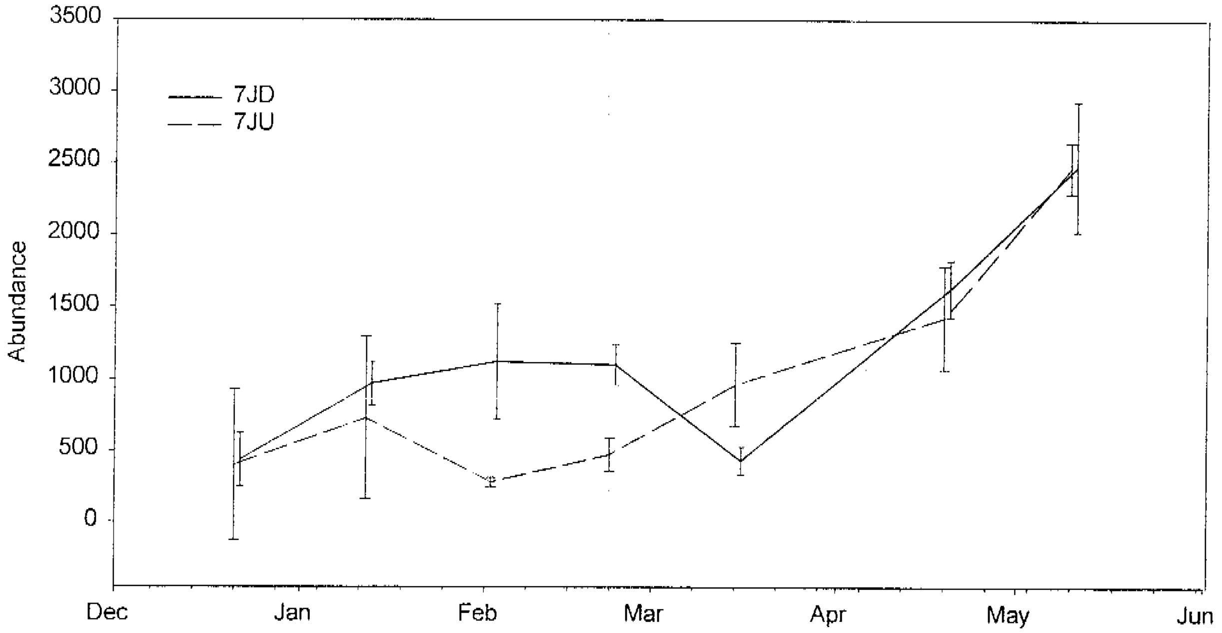


Figure 3.31 Taxa abundance at both sites in 7J over the wet season

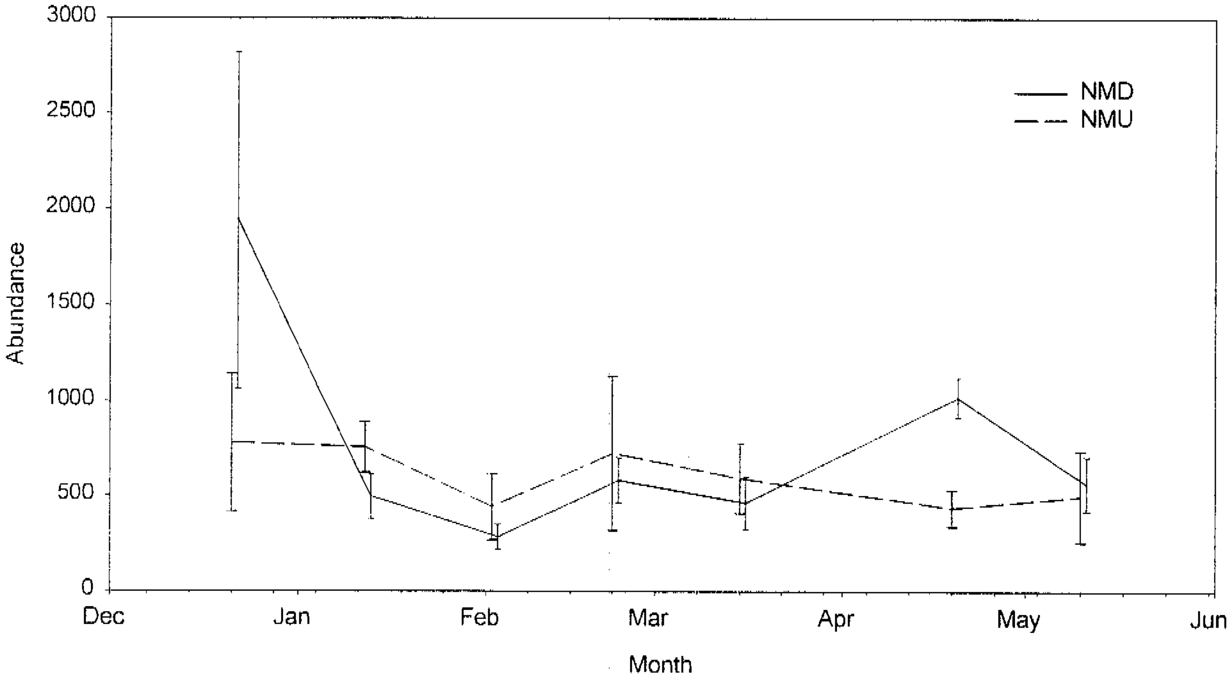


Figure 3.32 Taxa abundance at both sites in North Magela over the wet season.

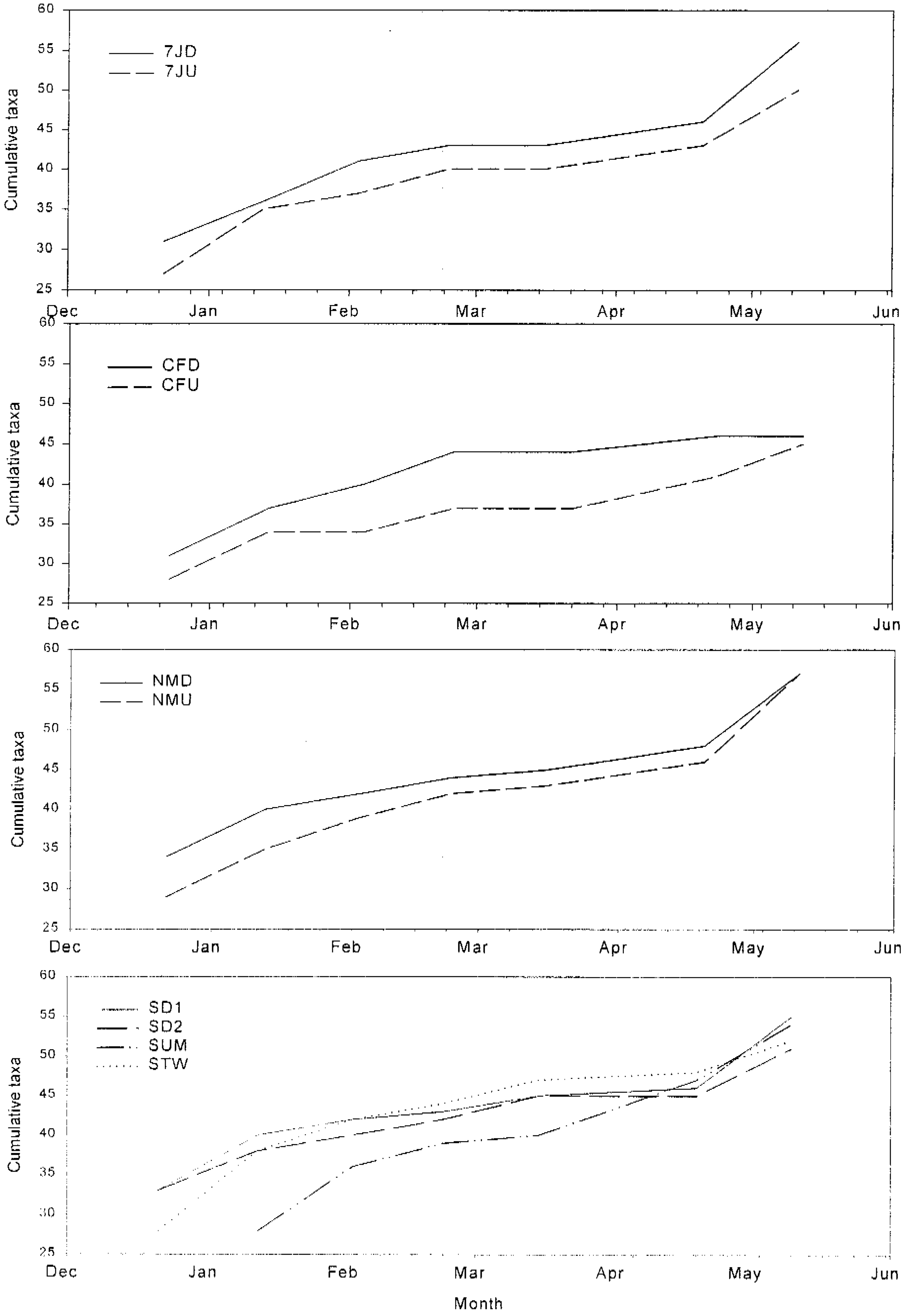


Figure 3.33 Taxa accumulation at all sites and all streams over the wet season

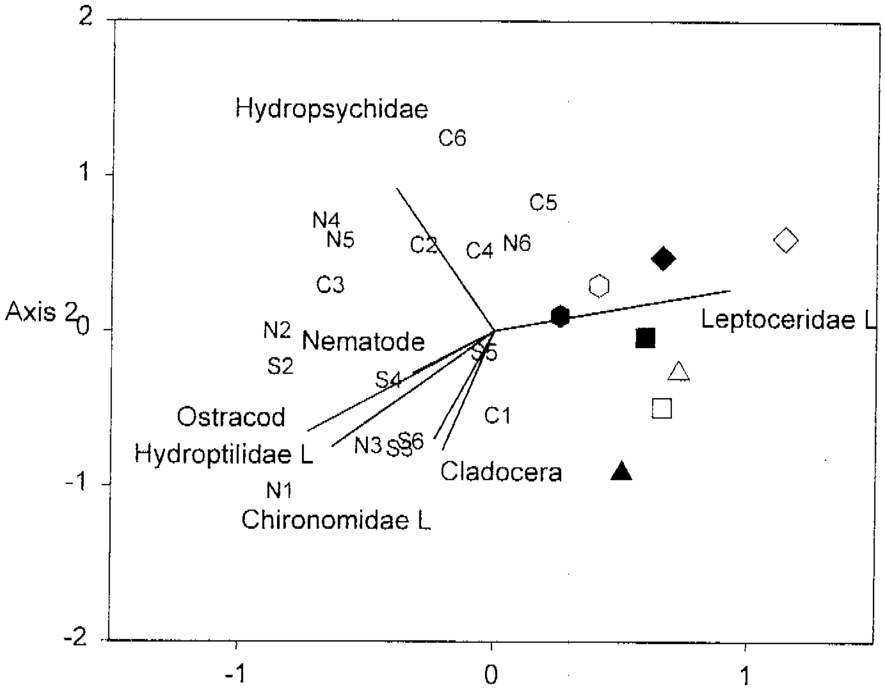
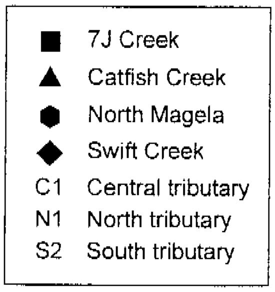


Figure 3.34 SSH-MDS ordination in 3 dimensions of tributary sites and main creek sampling sites from the sixth sampling event. (Stress = 0.17). Tributary numbers 1 - 6 indicate sampling position along creek. Significant taxa (p<0.01) are shown. (D/S sites are black and U/S sites are clear).



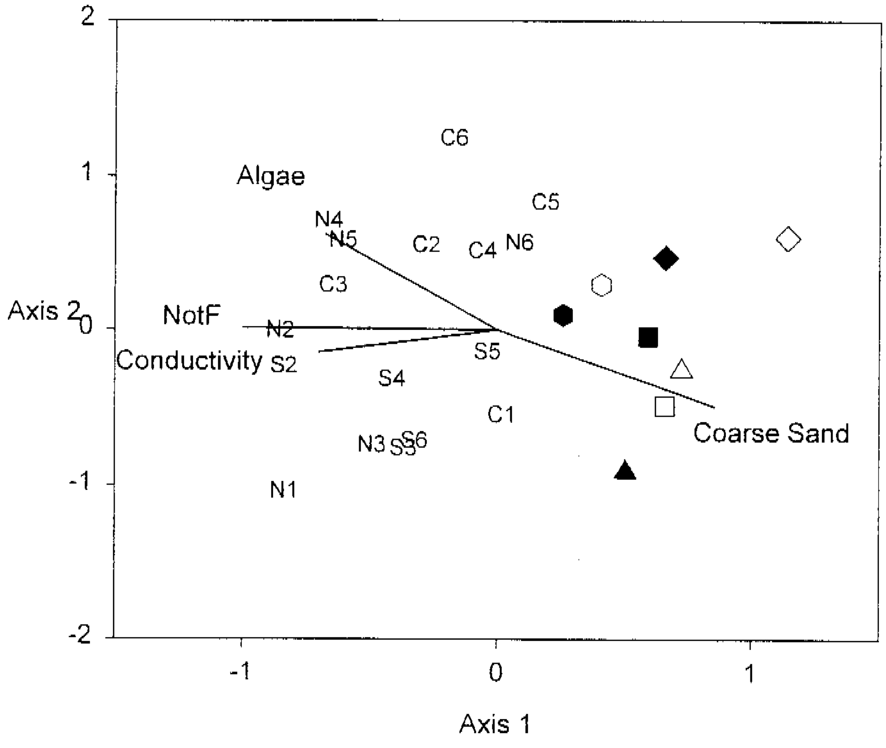


Figure 3.35. SSH-MDS ordination in 3 dimensions of tributary sites and main creek sampling sites from the sixth sampling event. (Stress = 0.17). Significant (p<0.01) environmental and habit variables are shown, except for Coarse Sand (p<0.02) and Algae (p<0.05).

and *Eriocaulon caespitosissima*) and Conductivity (both at $p<0.01$), Coarse sand ($p<0.02$) and Algae floc ($p<0.05$).

The ordination showed a separation of sites on the basis of tributaries versus main creeks. The significant variables on the ordination indicate that much of the difference was due to algae growth, and higher conductivity, found predominantly at the top North tributary sites (see Appendix 6). To a degree, we would expect some separation of sites, as the tributaries are all much smaller streams, thereby being influenced by different catchment processes. South tributary was largely unaffected by mine run-off with its source being some 200 - 250m southwards of the mine. Nevertheless, the sites along this tributary clustered together with the other tributary sampling locations, indicating a similarity based on the geomorphology of these streams. The ordination was most influenced by the top three sites along the North tributary.

Taxa which correlated significantly in the direction of the conductivity vector were Ostracods, Nematodes and Hydroptilidae larvae, and in a negative direction were Leptoceridae. Site conditions at the three uppermost North tributary sites especially, were exceedingly "muddy" and conditions appeared very similar to those of a backwater billabong. Flow along the North tributary fed into an underground seep for several hundred meters and it is possible that the water "ponded" along the top part of this tributary once the heavy wet season flows subsided.

Chapter 4: Discussion

Impact assessment studies are undertaken on the effects of a wide variety of perturbations which may be either natural, such as a forest fire, or anthropogenic, such as the construction of a mine. Regardless of their source, perturbations such as these all have the common element of being unreplicated and non-randomized which complicates the use of inferential statistics (Beyers 1998). The analytical short-comings of such situations, however, can be minimized by the use of multiple controls (Beyers 1998, Keogh and Mapstone 1995) and temporal replication (Stewart-Oaten et al 1986), as was the case in this study. However, the authors mentioned above, together with several others, have suggested that impact assessment may require long-term baseline data sets to be able to effectively discriminate between natural environmental variation and human induced effects (Faith et al 1990; Thrush et al 1994; Bunn 1995; Osenburg et al 1994). While this clearly, makes intuitive sense, such inter-annual time frames may simply not be an option when short-term impact assessment is required.

Studies for assessing the impact on the environment from development projects are often undertaken over periods as short as one year, or even over one season as was the case with this study. Faith et al (1990) argue that this lack of temporal replication reduces the statistical power of a study and represents the main challenge in the design of a monitoring program. In order to increase the certainty of results for short-term data sets, it is essential to use statistics in conjunction with a strong causal argument, which includes discussion of biological plausibility, biological gradients and consistency with previous studies (Hill 1965 cited in Beyers 1998). The present study has used a multi-facetted approach to statistical analysis based on the MBACIP design (Faith et al 1995) of rigorous hypothesis testing and inferential analysis, combined with causal argument to assess the potential impacts from the construction of the Jabiluka mine.

The causal argument focuses on the turbidity levels measured in Swift Creek. A comparison of these levels with earlier findings highlights their generally low levels and lack of persistence over the wet season. This information corroborates the findings of the rest of the study, leading to firm conclusions on the effects of mine construction on the macroinvertebrate communities in Swift Creek, downstream from the Jabiluka project area.

4.1Turbidity in Swift Creek

Turbidity levels were used in this study to estimate suspended sediment concentrations. Turbidity readings taken at the ***eriss*** gauging station located at the Swift Creek D1 site showed that several times within the first 6 weeks of sampling, the daily average remained steadily high and above 20 NTU. However, when values were averaged over a month they were much lower, with monthly values of approximately 13, 10 and 6 NTU in Swift Creek during the three months of highest turbidity for the wet season. These low averages indicate that the peak events recorded in Swift Creek at the downstream 1 site (SD1) may have occasionally reached high values, however, they were very short-lived events, generally lasting between several hours and a day. Similarly, in a Californian study, Harvey (1986) observed no apparent effects on macroinvertebrate communities at maximum levels of 50 NTU, against a background of 5 NTU, from suction dredge effluent. The periods of high, suspended sediment were short-lived events and concentrations showed a wide fluctuation (Harvey 1986).

In contrast to these previous findings, Stowar (1997) reported effects on the macroinvertebrate community 200 m below a stream crossing between July and September of the recessional flow period in Jim Jim Creek, Kakadu. During these three months, the stream crossing resulted in monthly turbidity averages of 20, 60 and 40 NTU above background levels. Paired upstream (control) - downstream dissimilarity values for these impacted communities remained high and sat outside the 95% confidence interval in a regression analysis with stream discharge. Marginal effects were still noticed 1 km downstream from impact where turbidity levels were 30 NTU above background.

The persistent increase in turbidity observed in Jim Jim Creek contrasts with the more sporadic, high stream discharges and associated spikes in turbidity recorded during the early part of the wet season in Swift Creek. Several studies have found that suspended sediment concentrations in savanna streams can show dramatic variation over the early wet season.

In Magela Creek, Northern Territory, Hart et al (1982) found a strong correlation between discharge and suspended sediment concentrations. A large total load was carried within a very short time frame and concentrations peaked some 8 hours prior to peak storm flow. This suggests that the first run-off waters to enter the stream carry most of the erodable material (Hart et al 1982). As Campbell and Doeg (1989) point out, weekly sampling is likely to miss these rising stages of flood events, (and 3 weekly even more so).

Fire is a widespread occurrence in northern Australia, and can cause variation in suspended sediment loads entering streams, irrespective of their hydrology and rainfall pattern over a wet season. Sediment concentration in two streams in Kapalga, Northern Territory, were found to be 10-100 times greater during storm events at the beginning of the wet season, than towards the end of the wet season, showing highest values for the stream burnt late in the dry season (Townsend 1997). For this stream, total suspended solids were found to be significantly higher (1600mg/L), approximately double the concentration found in the other stream (Townsend and Douglas 2000; Townsend and Douglas 1997). Therefore, some of the factors which affect the concentration of suspended sediment in streams are, amount of rainfall, where and when it falls in a particular catchment, length of time since last flood event, size of the event (Campbell and Doeg 1989), and the fire regime of the previous year. All these factors cause suspended sediment concentrations to fluctuate over short time intervals, on small spatial scales.

The fluctuating turbidity levels in Swift Creek at the SD1 site can be seen to be a response to localised discharge over the wet season, and were not persistent enough to disrupt macroinvertebrate composition at the community level. However, higher turbidity was recorded for the SD1 site than for the SUM site, for the first six weeks of sampling. The only factor that could account for this difference is the sediment input from the tributaries entering Swift Creek above the SD1 site. Their potential influence will be discussed in the following sections.

4.1.1 Potential influence of the mine tributaries and the eastern tributary (STE)

The results indicated a possibility that the Eastern tributary (STE) contributed to the sediment loads in Swift Creek, as turbidity was found to be relatively high in this stream over the wet season. However, significantly higher levels were recorded for both the tributaries draining the mine, North and Central. Visual observation in the field noted substantial sediment entering Swift Creek from the North tributary, located close to the SD1 site, early in the sampling program. Further work is needed to determine the exact contribution from each of the streams draining into Swift Creek. However, sediment clearly drained from the mine along North and Central tributaries, contributing to the higher turbidity values at the downstream site over the early sampling period.

4.1.1.1 Physico-chemical information on the mine site tributaries: North and Central.

The two main tributaries draining the Jabiluka mine site, North and Central, both received sediment loads from areas surrounding the mine during construction. Waters entering Central tributary potentially transported sediment from waste-rock stockpile run-off, road-crossings surrounding the mine, and cleared woodland. Waters entering North tributary had the potential to transport sediments from waste-rock used to line upstream drains near the dam wall, dam wall runoff and nearby cleared woodland.

Waste rock, and the blast residues it may contain, can provide a source of bio-available nitrate-nitrogen and ammonia-nitrogen to the surrounding tributaries (R. Batterham, EWLS, pers comm). The waste rock stockpiled during construction contained 0.945 tonnes of available soluble nitrogen and it was estimated that 75% of this was leached from the waste rock during the wet season flows of 1998/99, potentially contributing to downstream concentrations of nitrate and ammonia (at the D1 gauging station) three to four times higher than the pre-disturbance load from the catchment (R. Batterham, EWLS, pers. comm).

Nitrate levels were not monitored in this study, therefore no comparison can be made between Swift Creek and the control streams. The dry season naturally allows for the build up of nutrients within leaf litter and soils, often resulting in high initial concentrations of solutes, in particular nitrate, following the first flush of river catchments (Mitchell et al 1997). It is plausible there is a connection between increased nutrient levels in Swift Creek and high abundance of the filter-feeding Simuliidae in December, however, there is no direct evidence (nutrient data, increased algal biomass, other taxa responses) to support this.

4.1.1.2 Abundance of Simuliidae over the wet season

At the first downstream site in Swift Creek (SD1) recolonisation of Simuliidae was rapid and abundant early on in the wet season. Following the December sampling, a marked decline was observed, and recolonisation then resumed after a period of somewhere between 3-6 weeks, by the fourth sampling event. Simuliidae larvae require clean, unscoured sites for attachment (Williams and Williams 1993; Mackay 1992) and small increases in suspended solids have been shown to increase drift rates of both Chironomidae and Simuliidae, particularly for Simuliidae when concentrations reached 100 mg/L (White and Gammon 1977). It is possible that this level of suspended sediment was reached in Swift Creek, although it would not have been a prolonged event. Simuliidae do not seek refuge in the hyporheic zone, but rather, initiate drift as a response to unfavourable conditions. This means that they must re-establish from upstream sources, or other nearby perennial pools (Paltridge et al 1997).

A study undertaken in the River Necker, a Swiss pre-Alpine river, found Simuliidae to be very fast recolonisers, attaining undisturbed population levels within 1-6 days post disturbance. Drift was found to be the most important pathway of recolonisation in this particular study (Matthaei et al 1996). (Matthaei et al 1997) experimentally disturbed patches of this same stream over the following summer, and found a reduction of nearly 100% for Simuliidae, post-disturbance, with a lag phase of 12 days observed before recolonisation. (This was suggested to be a long lag phase due to the distance to source fauna for recolonisation). If Simuliidae are fast recolonisers, as earlier studies would seem to indicate (Matthaei et al 1996, 1997), then it appears that some factor or factors were responsible for making the habitat at SD1 unsuitable over a period of at least three weeks, possibly up to six weeks. Within the framework of this study it is impossible to attribute any changes to the Simuliidae population at the Swift downstream sites to mine construction alone. However, given that these taxa did not recolonise at the SD1 site for this established period of time, the possibility of mine influence cannot be ruled out.

4.2 Statistical analysis of paired-site dissimilarity and *difference* data for the four streams over time

Results of paired-site dissimilarities and taxa richness difference data both showed no significant differences between Swift Creek and the three control streams, either at the level of "Creeks overall", or for any particular interactions. Total abundance difference data also showed that Swift Creek produced values within a statistically similar range to two of the control streams, 7J and North Magela creeks, at both the Creek and Creek vs Time level of interaction. (Catfish Creek had a notably higher abundance at both sites over the entire sampling period and will be discussed in the following section). These results all provide firm evidence that measures of paired-site difference for Swift Creek were well within the environmental range presented by the Control streams. These measures were not only visually similar in time series analysis, but also statistically similar in ANOVA.

The dissimilarity values generated between the main upstream and downstream sites in Swift Creek were relatively low overall and were well within the range of values from the three control streams. An exception was the final sampling event where relatively large dissimilarities were produced between SUM and SD1; however, this difference was not found to be statistically significant based on the Tukey HSD test.

The SUM site was distinct from the downstream sites in containing stands of gallery forest dominated by *Allosyncarpia ternata* producing more shaded conditions that allowed greater abundance of detritus to accumulate as a consequence of increased leaf fall and tree roots trapping materials, and less attached algae because of reduced light. Grazers have been found in very low numbers or may be absent altogether from detrital aggregations, largely as a result of the unsuitability of this substrate for the proliferation of algae, their main food supply (Dobson 1991). The main upstream site in Swift Creek (SUM) showed a decline of the Ephemeropteran grazers- Baetidae and Caenidae- commencing around the 5th sampling event in March, leading to very low numbers by the last sampling occasion in May. The conditions of high detritus cover noted at the SUM site towards the end of the sampling period may have contributed towards a declining population of mayflies.

The relative lack of Ephemeropterans created a large dissimilarity in the final sampling between SUM and both downstream sites in Swift Creek (SD1 and SD2) where these taxa still formed part of the dominant fauna. Conditions at SUM for the final sampling event consisted of very low water levels (5-6 cm) and high detritus cover relative to the other sites. The recessional flows also brought on a greater dominance of Hemipterans and Coleopterans. These taxa may have gained prominence earlier at the SUM site as a result of reduced water level and flow rate. Grazers, such as Baetidae, may not compete well against these predatory and detritivorous species, as noticed by Dobson (1994), particularly given the environmental conditions at the SUM site at this time.

It is noteworthy that even with these contrasting environmental conditions between SUM and SD1 on the final sampling event, the faunal communities did not show up as significantly different. The statistical sensitivity of the Tukey HSD test may not have been high enough to detect the difference between SUM and SD1 on the 7th sampling event.

Difference in paired-site total abundance varied significantly between streams. However, all of these differences were caused by Catfish Creek, a more perennially-flowing stream where taxa abundance was higher relative to the other controls and Swift Creek. This is largely due to the contribution from Acarina and Oribatidae (water mites) and is discussed further in section 4.3.2.

The trend in abundance at all sites in Swift Creek was similar to that noted by Boulton and Lake (1992b, c) where peak abundance occurred just after the onset of flow, and again, before the cessation of flow (with the exception of SUM). The high abundance early on can be seen as a response to the initial flooding of the area, initiating an ecological chain of events, from release of nutrients to increases in primary and secondary production (Paltridge et al 1997). Subsequent flooding can then be responsible for a decline in macroinvertebrate abundance. Douglas (1999) found low richness and abundance during the first month of flow, following high initial levels in the pre-flow period. This decline was attributed to flood spates and their associated disturbance.

These results indicate that the trends in macroinvertebrate community structure within Swift Creek were within the natural range for the area over the 1998/99 wet season. The inclusion of three control streams strengthens this argument by increasing the inferential power of the statistics (Humphrey et al 1995). It is highly unlikely that all three controls would undergo changes to macroinvertebrate community structure and produce trends similar to Swift Creek as a result of some natural perturbation that coincided with mine construction. The use of paired sites and *difference*/dissimilarity data reduces the natural environmental variability of the region down to differences between stream "behaviour" and not isolated sites. It must be concluded that the observed patterns of *difference* between upstream and downstream sites for abundance, richness and community structure represent natural seasonal fluctuations and not an influence of the mine.

4.2.1 The relationship between stream discharge and paired-site dissimilarity

Stowar (1997) in the study on Jim Jim Creek, found a positive correlation between declining flow rate and dissimilarity over the dry season where macroinvertebrate communities became "more similar" as flow was reduced. Similar trends have been observed in the upper South Alligator River in Kakadu, where it has been noted that taxa richness increases over the recessional flow period in this permanently flowing stream, and upstream-downstream communities become more similar (***eriss***, unpublished data). Outridge (1988) observed a decline, overall, in taxa richness over the dry season in the seasonally-flowing Magela Creek, and a corresponding increase in abundance of the dominant taxa. In the case of Stowar (1997) these were predominantly the Chironomids. A recent study in the Nyamweru River in Uganda found a correlation between discharge and taxa abundance with low discharge correlated with high abundance during the dry season (Tumwesigye et al 2000). The dominant taxa in the Ugandan stream were also Chironomidae (>40%). As flow declines, so too does the catastrophic disruption to community structure, allowing the dominant dry season taxa to proliferate.

If decreasing dissimilarity over the dry season due to decreasing stream discharge is a general trend (see Stowar 1997), then the apparent trend over the wet season should be one of negligible to small fluctuation, if stream discharge, on average over the life cycle of invertebrates, does not display any obvious pattern and trend. Paired site dissimilarities remained reasonably constant over the wet season; the exception to this was Catfish Creek on the 4th sampling event, where a flood in the lower East Alligator catchment reduced abundance at the downstream site, providing an example of the increased "patchiness" that can be associated with local spates. It seems reasonable to postulate that any impact from sediment disturbance (if any) was too small to be detected, within the range of community dissimilarity observed over the wet season.

4.2.2 Conservation issues and the choice of data analysis

When the issue of concern in biological monitoring is maintenance of biodiversity at the level of community structure, then multivariate data is of prime importance, reflecting the response of a whole community of organisms (Faith et al 1995; Humphrey et al 1995). Smith et al (1988) propose that different species within a multivariate data set act as replicates of each other's responses, thereby diminishing the variability or "noise" in the data and, as a result, enhance statistical power. In an experimental study in a polluted stream, Faith et al (1995) found that the most sensitive response variable in a BACIP design was the dissimilarity value for the total community response - as opposed to abundance data for the most sensitive species. This contrasts to Underwood's (1993) finding on purported greater sensitivity of population measures. However, Underwood (1993, 1994) acknowledges that an impact may alter temporal or spatial variability of a population, and not just mean abundance.

In the absence of pre-impact data, and any findings of impact, it is not possible to determine which measure of paired-site difference was the most sensitive. The univariate and multivariate data together provide powerful corrobative evidence. Clearly, all three measures were needed to fully explain ecological patterns of faunal succession and to corroborate the findings of no-impact from mine construction.

4.2.3 Summary of mine influence

Suspended sediment in Swift Creek was found to be higher at the SD1 site than at the SUM site for the first 6 weeks of sampling. However, these levels of suspended sediment (as reflected by turbidity values) at SD1 were not consistently high over the wet season. Instead, there was much fluctuation evident, and average monthly values were relatively low. Therefore, it can be concluded that the levels of suspended sediment in Swift Creek downstream of the mine were not sufficiently high, or prolonged enough to have disrupted macroinvertebrate community structure over the 1998/99 wet season.

4.2.4 Potential reasons for not detecting an impact

When a conclusion of "no impact" is drawn, two scenarios are possible. The first is that the null hypothesis was correct and there really was no impact. The second possibility is that the null hypothesis was false, but the impact was not detected because of insufficient statistical power i.e. a Type II error (Faith et al 1990). Statistical power was increased spatially in this study through the incorporation of data from sites in three control streams. However, it is possible there was insufficient pre-impact temporal "replicates" to detect an impact. As the number of years (replicates) monitored prior to disturbance increases, the value for the maximum change or impact that would go undetected becomes smaller, for a given level of significance (Faith et al 1990).

Given the consistent seasonal trends of fauna succession (arch-shaped trajectories) in each stream, it is possible that the macroinvertebrate communities in the ARR region are well adapted to frequent spates and associated increases in suspended sediment over the wet season, as this forms part of a predictable seasonal pattern (see Resh et al 1988). The concentrations of suspended sediment transported during the high flow events for all the creeks may well have been within a comparable and tolerable range.

Factors that influence the rate of recovery after a wide variety of disturbances (logging, flooding, drought and chemical stressors) are: i) persistence of impact, ii) life history of the organism (including generation time, emergence time, and propensity to disperse, iii) time of year in which disturbance occurs, iv) presence of refugia and v) distance to source for recolonization (Niemi et al 1990). It is plausible that all of these factors worked to mitigate against obvious and sustained disturbance effects in Swift Creek and in the control streams. The construction of Jabiluka disturbed soil in the catchment above Swift Creek, however, suspended sediment washed into Swift Creek as a pulse event of short, intense peaks. The organisms are adapted to local conditions with short generation times (Marchant 1982b), and a high propensity to drift if required. The first flushes of the mine-site occurred at a time of year when it was difficult to distinguish it from the natural variability of the region, which can be high. Refugia may have been available within streams through a mosaic of patches left undisturbed from spates (Matthaei et al 1999). Finally, distance to a recolonisation source was readily available from upstream sources. Clearly, the potential to detect an impact (if one has occurred), from within the natural variability of the region, is reduced when factors operate in favour of the resilience of the fauna.

4.3 Seasonal trends in faunal succession for Swift Creek and the control streams

4.3.1 Temporal trends for all sites

The temporal trends displayed in ordination space for all streams in this study were "arch-shaped" seasonal trajectories. These were well defined for both Catfish Creek and Swift Creek (D1 vs both UM and TW). 7J Creek produced a more defined arched pattern for the downstream sites than the upstream ones, whereas North Magela showed the opposite pattern with better definition at the downstream site.

Catfish Creek maintained a tighter fit within this "arch", as did (but to a lesser extent) North Magela. This indicates a close biological spacing between sites over time (Clarke and Warwick 1994). Catfish is spring fed from the base of the escarpment for much of the dry season and the early resumption of flow in the wet season (relative to the other creeks) allowed for establishment of macrophytes and macroinvertebrate community structure prior to the beginning of the sampling period. When Boulton et al (1992a) studied a 2 year hydrological regime, they found that the dry stream phase (i.e degree of drying) had more influence on macroinvertebrate community assemblages than floods. The ability to survive the desiccation period appears to be a critical factor, setting the recolonisation pattern for the ensuing wet season (Boulton et al 1992b).

As described above, Catfish Creek flows for longer than the other streams in this study. The permanent nature of Catfish Creek, in the upper reaches, may therefore represent less of a "disturbance" over the dry season, giving certain taxa an advantage and decreasing colonising opportunities for other taxa. Permanent flow means that many species would be able to persist all year and thus the communities will change less dramatically than in more intermittent streams. This may explain the "tighter" fit within the arch-shaped trajectory displayed by Catfish Creek, where reduced room for opportunistic taxa means that the biological positions of the upstream and downstream sites were less "extended" than for the other streams.

The stability of the substrate is also a major factor in reducing the effects of flood spates on community structure (Cobb et al 1992; Dole-Olivier et al 1997). It is possible that Catfish Creek was less prone to disturbance from heavy rainfall events, especially at the upstream site because it had a well-established stand of macrophytes throughout the sampling period and these would stabilise the substrate.

The arch-shaped seasonal trajectories found in this study resemble those found in intermittently flowing streams by Douglas (1999) and Boulton and Lake (1992b). In studies of two seasonal streams in Victoria, Boulton and Lake (1992b) found that streams which flowed continuously produced a "doughnut shaped" trajectory in ordination space, representing a seasonal regression back to a similar biological community. Streams experiencing a dry period, produced "arched" patterns such as those observed in this study, indicating a break in flow conditions.

4.3.2 Main determinants of ordination space

Sites in ordination space were separated according to three significant correlates. Axis 1 produced a spread amongst the sites relating to changes in faunal composition over time. Those sites that were highly seasonal by nature, such as found in Swift Creek, spread far across the right-hand side of ordination space indicating that peak succession was reached around the fifth sampling event, and then faunal composition began the return to early/late flow conditions. Axis 3 produced a site separation explained by water flow, and the relationship between this axis and axis 1 was responsible for the arch-shaped trajectories observed in the streams over the wet season.

Paltridge et al (1997) in a study of early recolonisation patterns (first month) between lentic and lotic habitats, found that, with increasing time after commencement of flow, the faunal composition from sites in the seasonally-flowing stream tended to resemble more closely the fauna from the source sites. Source sites had a high taxa abundance as a result of permanent water and a diversity of micro-habitats. As this fauna drifted downstream, the temporary stream was colonised, creating similar communities to the upstream source. This trend observed by Paltridge et al (1997), was also seen in this study, where the upstream pattern was "followed" by the downstream pattern (external arches in the concentric pattern) over the wet season, indicating that similar communities were forming at the downstream sites relative to the upstream sites over time.

In contrast to the highly seasonal spread observed for Swift Creek, the Catfish Creek sites were situated within a smaller ecological range, to the left of the ordination space. The changes in faunal composition (which relate to the changing flow conditions over time) were more apparent in the temporary streams than in Catfish Creek. These sites did not differ markedly in faunal composition over the wet season compared with the intermittent streams, which contained a greater range of species (approximately 10 families more than Catfish Creek). Generally, perennial streams tend to have a greater taxa richness than intermittent streams (Boulton and Suter 1986). However, this was not the case for Catfish Creek.

This stream appears to offer a reduced opportunity for colonisation, possibly due to a hydrological regime that is intermediate between seasonally-flowing and permanently-flowing (along its length) streams. At either end of the continuum (permanent vs intermittent) colonisation opportunities exist for the taxa adapted to these extreme conditions of flow. With respect to intermittent streams, Paltridge et al (1997) acknowledge that different life history strategies of the fauna lead to variation in the timing of recolonisation, and a dynamic successional process. During the early wet season flows in intermittent streams, large areas of aquatic habitat suddenly become available, together with the addition of fresh leaf litter and detritus which flushes into the streams. This provides an abundant food and energy source and habitat area. The life-history strategies of the early colonists have adapted to take advantage of these conditions (Tripodi 1996). Similarly, with the dry season, pool formation creates opportunities for colonisation by a different lentic fauna, while life-history characteristics have evoloved that enable the taxa to survive, or re-emerge at the beginning of each season.

The constantly changing physico-chemical conditions of an intermittent stream therefore encourage a variety of taxa (Paltridge et al 1997). Permanently-flowing streams tend to have high taxa richness as a result of the diversity of microhabitats found (Boulton and Suter 1986; Paltridge et al 1997). Catfish Creek, situated "halfway" between these situations possibly demonstrates less physico-chemical and flow-related fluctuation over the wet season, thereby attracting a less diverse range of macroinvertebrate fauna.

A dominance of the water mites, Acarina and Oribatidae, at Catfish Creek, may also contribute to the observed trends. Most species of water mites have a wide pH tolerance (A substantially lower pH was found at the Catfish Creek sites relative to all the other sites). They have also been found to be parasitic in their larval stages on a wide range of insect orders including; Collembola, Odonata, Hemiptera, Coleoptera, Diptera and Trichoptera (Di Sabatino et al 2000). It is speculative, however possible, that the high abundance of water mites found at Catfish Creek constrained community succession in some way, either through occupying much of the available habitat niches in the benthos, or through parasitism on other taxa. This speculation clearly requires further work.

Axis 2 defined a gradient relative to macroinvertebrate abundance, and also distance from the recolonising source of taxa, the headwaters, which Paltridge (1992) found was the main source of recolonisation in seasonally-flowing Magela Ck in the ARR. Catfish Creek sites were located in close proximity to the headwaters (~2 km from the upstream site) producing the highest abundance at both sites over the wet season. North Magela sites were located furthest from the headwaters (11 km from the upstream site) and showed the lowest abundance overall. Swift Creek and 7J sites sat approximately the same distance from their headwaters (~2.8 km from the upstream site for both) and produced comparable taxa abundance for all sites. Paltridge et al (1997) did not find distance from source to be a significant factor in Magela Creek mainly because there was not sufficient spatial range incorporated in their study.

The three most important correlates of ordination space; water flow, the seasonal range of taxa succession, and distance to recolonisation source, account for the seasonal trends found in this study (as discussed in section 4.3.1 and 4.3.2). It is apparent that Swift Creek more closely resembles the two control streams located in the Magela Creek catchment (7J and North Magela creeks), and Catfish Creek occupies a distinct biological position.

4.3.3 Fauna recolonisation and succession over the wet season

Different flow periods encourage a diversity of taxa. Some families require high flow conditions such as the filter feeders Simuliidae and Hydropsychidae, that attach to the substrate and take nutrients from the water column. Other taxa dominate when flow ceases and residual surface waters become stagnant. For example, predatory taxa such as Coleopterans and Hemipterans increase as flow ceases in response to large numbers of taxa stranded in pools (Dobson 1994). Results indicate underlying similarities in patterns of recolonisation and succession between Swift Creek and the three Control creeks. Much temporal and spatial variability was evident between sites. However, the major taxa involved in the seasonal changes over the wet season were remarkably similar in all the streams (apart from Catfish Creek for water mites). Of the four major pathways of recolonisation from dry season refuges; drift, migration from downstream, vertical migration and aerial dispersion (Williams and Hynes 1976), the most important one operating in the ARR is drift from upstream perennial reaches or other permanent water sources such as billabongs (Gunn 1997; Paltridge et al 1997). However, recolonisation from the sediment (vertical migration) can also be an important source (Paltridge et al 1997).

This study found the dominant early recolonisers to be Oligochates, Nematodes, Microcrustaceans and water mites at all sites within the four streams. These taxa are not dependent on food materials growing on the substrate and hence can be opportunistic early on in unoccupied areas (Mackay 1992). They all have dessication-resistant eggs and can hatch directly from the sediment; the dry sandy substratum of the Magela Creek bed was found to provide a significant refuge for these taxa (Paltridge et al 1997). Cladocerans and copepods were the dominant early colonisers in the Kapalga streams studied by Douglas (1999), and were observed in pools 24 hours after wetting. These taxa showed a remarkable ability to avoid desiccation over a long dry season period of 6-9 months (Douglas 1999). Similar results of early recolonisation by Oligochaetes, microcrustaceans and water mites were found by Paltridge et al (1997) and Boulton and Lake (1992a), these taxa hatching *in situ* or being present in the very early drift from nearby perennial sources (Paltridge et al 1997). Leptophlebiidae were another early dominant taxa in this study, and as opportunistic detritivores, may have been responding to the high amounts of organic matter flushing in from the surrounding catchments with the first wet season flushes.

Early colonizers quickly gave way to taxa that contributed the most to population abundance in all streams until the recessional flow period began. The Baetidae and Caenidae mayflies were found in varying abundances by the second and third sampling event in Swift Creek and each of the controls, most abundantly at Catfish

Creek, with the exception of SUM. They were particularly dominant at Swift Creek for the final third of the sampling program, especially, with the exception of SUM. They were also the "core" taxa found in Victorian intermittent streams (Boulton and Lake 1992a).

Trichoptera did not increase in numbers significantly until the last two sampling events for Swift, Catfish and 7J creeks. 7J also had substantial numbers of this taxon on the third sampling event at the downstream site. There were never high numbers of Trichoptera at either site in North Magela. This taxon was classed as a "mid-successional" colonizer by Boulton and Lake (1992a) and found to predominate along with the Ephemeropterans throughout the middle of the study period. Some Trichoptera such as Hydropsychids need rougher surfaces for attachment that may not be available until algal mass builds up on stones, partly explaining their later colonization habits (Mackay 1992). They are also detritivores and therefore rely upon the breakdown of organic matter (Tripodi 1996). This process may occur at a faster rate as water temperatures rise temporarily in the ponded areas which develop during receding flows, and may explain their later influence upon community structure (Hemphill and Cooper 1983).

The dominance of the later taxa is evident for all creeks as recessional flow commences with increasing abundance of Chironomidae, Hemipterans, Odonates and predatory Dytiscids, amongst other beetles. This was also the trend found by Boulton and Lake (1992a) and they describe these taxa as the "summer" taxa most commonly associated with lentic waters. Chironomids increased in numbers for the last two sampling events in Swift Creek and slowly over the sampling period for 7J and Catfish Creek. High numbers were found in North Magela at the beginning and end of the sampling regime.

The multivariate technique of ordination has proven to be a highly useful tool in the analysis of community data sets providing for a comparison between temporal patterns of taxa succession. This study seems to support the contention of Boulton and Lake (1992b) and Douglas (1999) that there is an underlying seasonal periodicity to intermittent streams, regardless of the abiotic and taxonomic variation that occurs between different geographical regions. Trends in faunal succession over the wet season that have been found in Magela Creek (Tripodi 1996; Paltridge et al 1997) also now appear to apply to the streams in the ARR on a more regional basis.

4.4 Limitations to the study

The main limitation of no pre-impact data has been mentioned previously, and discussed with respect to modifications of the MBACIP design (see section 1.3.2).

4.4.1 The Eastern Tributary

The MBACIP design employed in this study is based upon the comparison between upstream and downstream sites from within the same creek, amongst a range of different creeks, including several (three) controls and one potentially-impacted creek (Faith et al 1995). Creeks are compared for patterns of temporal change based on a *difference* value produced between the up and downstream sites. This reduces the natural variation found at different sites to "within" creek biological differences and a major premise of this design is that similar lotic influences are acting upon both the up and downstream sites. From this premise the MBACIP design can infer impact if the treatment creek (Swift Creek) lies outside of the range presented by the control creeks. However, this design also assumes that there should be no other confounding variables that may influence the sites. When selecting sites from within a natural environment, it may be difficult to hold constant all other lotic influences.

Swift Creek receives inflow from a number of tributaries that intercede between the up and downstream sites. Three of these are the tributaries draining the mine site from the western side; North, Central and (to a lesser degree) South tributary. A tributary also enters Swift Creek from the eastern side draining into the creek above the downstream sampling sites. This eastern tributary (STE) is a younger channel to the other tributaries and has been found to contribute large sediment loads into Swift Creek (K. Evans, ***eriss***, pers. comm). It was found to be the most turbid of all the creeks monitored in the Swift Creek catchment area during the 1998/99 wet season. The Eastern tributary introduces an uncertainty into the analysis. Until its dynamics are well understood, it reduces the inferential power of the study which attempts to exclude all other possible causes of an observed change, leaving the only explanation to be the impact in question, i.e. mine construction (Humphrey et al 1995).

4.4.2 Level of Taxonomic resolution

Family level identification in this study was able to discriminate well between sites for patterns of seasonal community structure changes, producing arch-shaped trajectories for Catfish Creek, Swift Creek and to a lesser degree, 7J Creek and North Magela. This particular pattern is known to be representative of intermittent streams in tropical and temperate regions. The loss of the family Simuliidae was evident from the SD1 and SD2 sites after the first sampling event in December. However, more subtle community-level changes were not noted and it is possible that species level information would have revealed more information.

The family Chironomidae have several species that are particularly sensitive to suspended sediment (P Cranston, pers. comm). Stowar (1997) found that Chironomidae were adversely affected by increased suspended sediment from a road crossing in Kakadu over the dry season. This effect in Jim Jim Creek was observed using family level identification., However, the more subtle turbidity levels recorded in this Jabiluka study suggest that a more sensitive approach (higher taxonomic level of identification) may have improved the discriminatory power.

4.5 Recommendations for Future Monitoring

4.5.1 Monitoring of the Tributaries

The tributaries draining the mine site (North and Central) would provide the first indicators of any disruption to the benthic community resulting from mine activities. North tributary provided evidence of being the site most influenced by mine construction indicated by high conductivity values recorded during late wet season sampling, with values ranging from 19.5 - 50 µS/cm. Corbett (1996) and O'Connor et al (1995) both found that conductivity separated out mine influenced waterbodies from undisturbed locations, suggesting there is some correlation to human disturbance. High conductivity has also been associated with other sites (mainly billabongs) in the ARR that have been disturbed by mine activities (OSS 1993). An earlier study in the Magela Creek catchment (which encompasses Swift Creek) found a range for conductivity between 10-21 µS/cm indicating the naturally low levels of dissolved salts prior to mine construction (Hart et al 1982). In accordance with the "precautionary principle" for environmental impact assessment, it would seem wise to continue monitoring the three uppermost sites at North tributary during the late wet season sampling, as part of the ongoing baseline monitoring for Jabiluka.

4.5.2 Sampling sites and Control streams

The SUM site in Swift Creek was heavily influenced by detritus, most notably towards the end of the sampling period, and this lead to large dissimilarity values for the last sampling event with the SD1 site. Even though no statistical differences were found based on these dissimilarities, it is important to be able to match habitats as closely as possible (in the treatment stream especially) in order to avoid any confounding results. If possible, a site along the main branch that has less detritus cover would be better for future monitoring.

The inclusion of a second upstream site in the treatment creek (STW) is not a necessary aspect of future monitoring (past the 2nd year of data collecting in 2000). The large biological distance, (at times statistically significant in relation to the control streams), shown by STW in relation to SD1 invalidates this site as a sound upstream control. Too much variation is introduced, producing large dissimilarity values that are the result of habitat differences rather than any mine-related factors acting upon the downstream site.

4.5.3 Level of taxonomic identification

Species level identification of Chironomidae larvae should be included as part of future monitoring so as to be able to assess if any particular species are being affected by sediment concentrations. As mentioned previously, several species of Chironomid are known to be particularly sensitive to elevated sediment concentrations (P Cranston pers. comm) and information on their abundance would increase certainty of conclusions.

4.5.4 Analysis of sediment in the tributaries

It is important to be able to separate influences from the mine from those of the Eastern tributary. Sediment from mine construction run-off could be expected to contain higher levels of organic material as a result of soil disturbance and potential erosion effects, in relation to the Eastern tributary. An assessment of suspended solids at the point of entry into Swift Creek for each of the three major tributaries; North, Central and Eastern, and also at the SD1 site, would establish the contribution of each to the total loads found downstream in Swift Creek.

4.5.5 Measurement of environmental variables

Swift Creek and two of the control streams, 7J and North Magela have gauging stations monitoring water level and turbidity data. One of the difficulties with this study was the inability to compare turbidity readings and stream flow between Creeks on anything less than a weekly interval. During the dry season, this time frame is acceptable given the lack of variability. Over a wet season, these variables alter rapidly and constantly and comparable readings (even on a daily basis) would provide an indication of localised conditions and form a basis for comparing creek discharge, which is a highly influential variable in determining macroinvertebrate community structure.

4.6 Conclusion

This study was undertaken over one season, in an environment characterised by extreme seasonal water level fluctuations. In the absence of pre-impact data, the strongest conclusions pertaining to impact must be inferred from a suite of inferential statistical analysis (e.g. ANOVA, time series and multivariate ordinations). From the conservation perspective, the advantage of the MBACIP design is that information about the ecological importance of any observed changes will be enhanced because data from the entire community of organisms are used (Humphrey et al 1995; Faith et al 1995).

The results indicate that Swift Creek was not noticeably disturbed by the construction of the Jabiluka mine. It is also possible that any disturbances that did occur were either too small, or too short-lived, to be detected within the fluctuating environmental conditions observed over the 1998/99 wet season. It could therefore be argued that any small change observed (i.e. temporary loss of Simuliidae at the downstream sites - even assuming this was mine-related) would not qualify as an important disturbance, in that macroinvertebrate community structure in Swift Creek was not perceptibly altered, either at any one particular sampling event, or over the wet season in general. Therefore, a strong causal argument based on these following points, leads firmly to a conclusion of no impact.

- Water quality (apart from turbidity) showed no anomalies in Swift Creek over the wet season

- Turbidity fluctuated over the wet season in Swift Creek, indicating that peak events of suspended sediment concentration were relatively low and not persistent.
- No other studies have reported effects on macroinvertebrate communities at similar levels of suspended sediment (i.e. relatively low and short-lived).
- Evidence from ANOVA based on multivariate and univariate parameters showed no significant differences between streams, apart from those attributable to high abundances in Catfish Creek.
- Seasonal trends in taxa succession, abundance and richness for Swift Creek fell within the natural environmental range and patterns displayed by the control streams.
- No biological gradient, showing changes to macroinvertebrate community structure, was detected at the SD2 site.

The inferential capacity of the MBACIP design is considered to be high with the inclusion of three control streams and paired site sampling within streams, especially when pre-disturbance data were unavailable. The ability to observe stream "behaviour" over a wet season is a very powerful biological tool, using the natural relationship that links up and downstream sites within the same stream to an advantage. It can be concluded that the biological monitoring program provided the mine with conclusive evidence of no-impact from the construction of Jabiluka, and clearly, biological monitoring programs have a very important role in impact assessment and monitoring.

References

ANCA (1999). *Kakadu National Park Plan of Management* 1999-2004. Kakadu Board of Management and Parks Australia.

ANZECC and ARMCANZ. (2000). Australian and New Zealand Guidelines for Fresh and Marine Water Quality, Column 1, *Australia and New Zealand Conservation Council. Agriculture and Agriculture and Resource Management Council of Australia and New Zealand*

Barton, B.A. (1977). Short term effects of highway construction on the limnology of a small stream in southern Ontario. *Freshwater Biology* 7: 99-108.

Batterham, R. (1999). Jabiluka Baseline Surveys: Benthic macroinvertebrates - Progress Report. *Progress Report to ERA Ranger mine.* ERA Environmental Services. Pty Ltd. August, Darwin, N.T.

Beisel, J.N., Usseglio-Polatera, P., Thomas, S., and Moreteau, J.C. (1998). Effects of mesohabitat sampling strategy on the assessment of stream quality with benthic invertebrate assemblages, *Archiv fur Hydrobiologie* **142**, 493-510.

Belbin, L. (1993). PATN Pattern Analysis Package v3.6. *Commonwealth Scientific and Industrial Research Organisation, Division of Wildlife and Ecology, Canberra, Australia PATN Pattern Analysis Package v3.6.* Commonwealth Scientific and Industrial Research Organisation, Division of Wildlife and Ecology, Canberra, Australia.

Beyers, D.B. (1998). Causal inference in impact studies. *Journal of the North American Benthological Society* **17**: 367-373.

Bisson, P.A. and Bilby, R.E. (1982). Avoidance of suspended sediment by juvenile coho salmon. *North American Journal of Fisheries Management* **2**: 371-374.

Blyth, J.D., Doeg, T.J., and St Clair, R.M. (1984). Response of the macroinvertebrate fauna of the Mitta Mitta River, Victoria, to the construction and operation of Dartmouth Dam. 1: Construction and initial filling period. *Occasional Papers of the Museum of Victoria.* **1**: 83-100.

Boulton, A.J. and Suter, P.J. (1986). Ecology of temporary streams: an Australian perspective. *In* De Decker, P. and Williams, W.D. (eds). C.S.I.R.O. and Dr. W. Junk. *Limnology in Australia*, 313-327. Melbourne.

Boulton, A.J. and Lake, P.S. (1992a). The macroinvertebrate assemblages in pools and riffles in two intermittent streams (Werribee and Lerderderg Rivers, southern central Victoria) *Occasional Papers from the Museum of Victoria.* **5**: 55-71

Boulton, A.J and Lake, P.S. (1992b). The ecology of two intermittent streams in Victoria, Australia. II. Comparisons of faunal composition between habitat, rivers and years. *Freshwater Biology* **27**: 99-121.

Boulton, A.J and Lake, P.S. (1992c). The ecology of two intermittent streams in Victoria, Australia. III. Temporal changes in faunal composition, *Freshwater Biology* **27**: 123 - 138.

Boulton, A.J., Peterson, C.G., Grimm, N.B. and Fisher, S.G. (1992a). Stability of an aquatic macroinvertebrate community in a multiyear hydrological regime. *Ecology* **73**: 2192-2207.

Boulton, A.J., Stanley, E.H., Fisher, S.G. and Lake, P.S. (1992b). Over-summering strategies of macroinvertebrates in intermittent streams in Australia and Arizona *In* Roberts, R.D. and Bothwell, M.L. (eds). NHRI Symposium Series 7. Environment Canada, Saskatoon.

Brewin, P.A., Buckton, S.T. and Ormerod, S.J. (2000). The seasonal dynamics and persistence of stream macroinvertebrates in Nepal: do monsoon floods represent disturbance? *Freshwater Biology* **44**: 581-594.

Bunn, S.E. (1987). Processing of leaf litter in two northern jarrah forest streams, Western Australia:II. The role of macroinvertebrates and the influence of soluble polyphenols and inorganic sediment. *Hydrobiologia* **162**: 211-223.

Bunn, S.E. (1995). Biological monitoring of water quality in Australia: workshop summary and future directions, *Australian Journal of Ecology* **20**: 220-227.

Campbell, I.C. and Doeg, T.J. (1989). Impact of timber harvesting and production on streams: a Review. *Australian Journal of Marine and Freshwater Research* **40**: 519-539.

Cao Y; Williams W.P. and Bark A.W. (1997). Effects of sample size (replicate number) on similarity measures in river benthic aufwichs community analysis, *Water Environment Research.* **69**: 107-114.

Chapman, M.G., Underwood, A.J., and Skilleter, G.A. (1995). Variability at different spatial scales between a subtidal assemblage exposed to the discharge of sewage and two control assemblages. *Journal of Experimental Marine Biology and Ecology* **189**: 103-122.

Chatfield, C. (1984). *The Analysis of Time Series: An Introduction.* 3rd edition, Chapman and Hall, London, UK

Chessman, B.C., Robinson, D.P., and Hortle, K.G. (1987). Changes in the riffle macroinvertebrate fauna of the Tanjil River, southeastern Australia, during construction of Blue Rock Dam. *Regulated Rivers Research and Management* **1**: 317-329.

Chessman, B.C. (1995). Rapid Assessment of rivers using macroinvertebrates: a procedure based on habitat-specific sampling, family level identification and a biotic index. *Australian Journal of Ecology* **20**: 122-129.

Chutter, F.M. (1968). The effects of silt and sand on the invertebrate fauna of streams and rivers. *Hydrobiologia* **34**: 57-76.

Clarke, K.R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* **18**: 117-143.

Clarke K.R. and Warwick R.M. (1994). *Change in Marine Communities: An approach to statistical Analysis and Interpretation.* Plymouth Marine Laboratory, Natural Environmental Research Council, UK.

Cline, L.D., Short, R.A., and Ward, J.V. (1982). The influence of highway construction on the macroinvertebrates and epilithic algae of a high mountain stream. *Hydrobiologia* **96**: 149-159.

Cobb, D.G., Galloway, T.D. and Flannagan, J.F. (1992). Effects of discharge and substrate stability on density and species Composition of stream insects. *Canadian Journal of Fisheries and Aquatic Sciences* **49**: 1788-1796.

Corbett, L. (1996). Aquatic Studies at Ranger Mine: A whole ecosystem approach, Final report. *CSIRO Division of Wildlife and Ecology,* Darwin.

Cuffney, T.F., Gurtz, M.E and Meador, M.R. (1993). Guidelines for the processing and quality assurance of benthic invertebrate samples collected as part of the National water-Quality Assessment Program. *U.S. Geological Survey, Report No 93-407*, 80 p.

Culp, J.M., Wrona, F.J., and Davies, R.W. (1986). Response of stream benthos to fine sediment deposition versus transport, *Canadian Journal of Zoology* **64**: 1345-1351.

Davey, G.W., Doeg, T.J., and Blyth, J.D. (1987). Changes in benthic sediment in the Thompson River, southeastern Australia, during construction of the Thompson Dam. *Regulated Rivers* **1**: 71-84.

Davies-Colley, R.J., Hickey, C.W., Quinn, J.M., and Ryan, P.A. (1992). Effects of clay discharges on streams. 1. Optical properties and epithelion. *Hydrobiologia* **148**: 215-234.

Davies, P.E. and Nelson, M. (1994). Relationship between riparian buffer widths and the effects of logging on stream habitat, invertebrate community composition and fish abundance. *Australian Journal of Marine and Freshwater Research* **45**: 1289-1305.

Death, R.G. and Winterbourne, M.J. (1994). Environmental stability and community persistence: a multivariate approach. *Journal of the North American Benthological Society* **13**: 125-139.

Di Sabatino, A., Gerecke, R. and Martin, P. (2000). The Biology and ecology of lotic water mites (Hydrachnidia) *Freshwater Biology* **44**: 47-62.

Dobson, M. (1991). An assessment of mesh bags and plastic leaf traps as tools for studying macroinvertebrate assemblages in natural leaf packs. *Hydrobiologia* **222**: 19-28.

Dobson, M. (1994). Microhabitat as a determinant of diversity - stream invertebrates colonizing leaf packs. *Freshwater Biology* **32**: 565-572.

Doeg, T.J., Davey, G.W, and Blyth, J.D. (1987). Response of the Aquatic Macroinvertebrate communities to dam construction on the Thomson river, southeastern Australia, *Regulated Rivers: Research and Management* **1**: 195-209.

Doeg, T.J., Lake, P.S., and Marchant, R. (1989). Colonisation of experimentally disturbed patches by stream macroinvertebrates in the Acheron River, Victoria. *Australian Journal of Ecology* **14**: 207-220.

Doeg, T.J. and Milledge, G.A. (1991). Effect of experimentally increasing concentrations of suspended sediment on macroinvertebrate drift, *Australian Journal of Marine and Freshwater Research* **42**: 519-526.

Dole-Olivier, M.J., Marmonier, P., and Beffy, J.L. (1997). Response of invertebrates to lotic disturbance- is the hyporheic zone a patchy refugium. *Freshwater Biology* **37**: 257-276.

Douglas, M. (1999). *Tropical savanna streams: their seasonal dynamics and response to catchment disturbance.* Doctoral thesis. Department of Biological Sciences, Monash University.

Environment Australia. (1996). Draft guidelines for an environmental impact statement on the proposed development of Jabiluka No 2 Uranium Mine. *Environment Protection Agency (Australia), and Northern Territory. Department of Lands, Planning and Environment.* Draft for public comment.

Extence, C.A. (1978). The effects of motorway construction on an urban stream, *Environmental Pollution* **17**: 245-252.

Fairchild, J.F., Boyle, T., English, W.R., and Rabeni, C. (1987). Effects of sediment and contaminated sediment on structural and functional components of experimental stream ecosystems. *Water, Air and Soil Pollution* **36**: 271-293.

Faith, D.P. and Norris, R.H. (1989). Correlation of environmental variables with patterns of distribution and abundance of common and rare freshwater macroinvertebrates. *Biological Conservation* **50**: 77-98.

Faith, D.P., Humphrey, C.L. and Dostine, P. (1990). Requirements for effective biological monitoring of aquatic ecosystems in the area of the Kakadu Conservation Zone. *Submission to the Inquiry into the Kakadu Conservation Zone.* Office of the Supervising Scientist. ARRI document 1652.

Faith, D.P., Humphrey, C.L. and Dostine, P.L. (1991). Statistical power and BACI designs in biological monitoring: comparative evaluation of measures of community dissimilarity based on benthic macroinvertebrate communities in Rockhole Mine Creek, Northern Territory, Australia. *Australian Journal of Freshwater Research* **42**: 589-602.

Faith, D.P., Dostine, P.L., and Humphrey, C.L. (1995). Detection of mining impacts on aquatic macroinvertebrate communities: Results of a disturbance experiment and the design of a multivariate BACIP monitoring program at Coronation Hill, Northern Territory. *Australian Journal of Ecology* **20**: 167-180.

Gardner, M.B. (1981). Effects of turbidity on feeding rates and selectivity of bluegills. *Transactions of the American Fisheries Society* **110**: 446-450.

Garmen, G.C. and Moring, J.R (1993). Diet and annual production of two boreal river fishes following clear-cut logging. *Environmental Biology of Fishes* **36**: 301-311.

Gauch Jr. H.G. (1982). *Multivariate Analysis in Community Ecology*. Cambridge University Press, Cambridge.

Gippel, C.J. (1988). The effect of Water Colour, Particle size and Particle Composition on Stream Water Turbidity measurements, Working Paper 1982/3. *Department of Geography and Oceanography*. University College, Australian Defence Force Academy, Canberra.

Gippel, C.J. (1989). The use of turbidimeters in suspended sediment research. *Hydrobiologia* **176/177**: 465-480.

Gray, L.J. and Fisher, S.G. (1981). Postflood recolonisation pathways of macroinvertebrates in a lowland Sonoran Desert stream. *American Midland Naturalist* **106**: 249-257.

Gray, L.J. and Ward, J.V. (1982). Effects of sediment releases from a reservoir on stream macroinvertebrates. *Hydrobiologia* **96**: 177-184.

Green, R.H. (1979). *Sampling Design and Statistical Methods for Environmental Biologists*. Wiley, New York

Growns, I.O., and Davis, J.A. (1994). Effects of forestry activities (clearfelling) on stream macroinvertebrate fauna in south-western Australia. *Australian Journal of Marine and Freshwater Research* **45**: 963-975.

Gunn, B. (1997). The macroinvertebrate colonisation of the tropically, seasonally-inundated, Magela Creek floodplain. *Supervising Scientist Internal Report* 250.

Hart, B.T., Davies, S.H.R. and Thomas, P. (1982). Transport of Iron, Manganese, Cadmium, Copper and Zinc by Magela Creek, Northern Territory, Australia. *Water Resources* **16**: 605-612.

Harvey, B.C (1986). Effects of suction gold dredging on fish and invertebrates in two California streams. *North American Journal of Fisheries Management* **6**: 401-409.

Hawking, J.H. (1995). *Monitoring River Health Initiative Taxonomic Workshop Handbook*. Murray-Darling Freshwater Research Center, Albury, Australia.

Hawking, J.H. and O'Connor, R. (1997). LWRRDC Final report: Quality assurance of identifications for the MRHI state/territory bioassessment program Internal Report 258, Supervising Scientist, Canberra and Darwin, Unpublished Report.

Hellawell, J.M. (1986). *Biological Indicators of Freshwater Pollution and Environmental Management*, Elsevier Applied Science Publishers, Essex

Hemphill, N. and Cooper, S.D. (1983). The effect of physical disturbance on the relative abundance of two filter-feeding insects in a small stream. *Oecologia* **58**: 378-382.

Hogg, I.D., and Norris, R.H. (1991). Effects of runoff from land clearing and urban development on the distribution and abundance of macroinvertebrates in pool areas of a river. *Australian Journal of Marine and Freshwater Research* **42**: 507-518.

Humphrey, C.L., Bishop, K.A., and Brown, V.M. (1990). The use of biological monitoring in the assessment of mining wastes on aquatic ecosystems of the Alligator River region, tropical northern Australia. *Environmental Monitoring and Assessment* **14**: 139-181.

Humphrey, C.L., Faith, D.P. and Dostine, P.L. (1995). Baseline requirements for assessment of mining impact using biological monitoring. *Australian Journal of Ecology* **20**: 150-166.

Humphrey, C.L. and Thurtell, L. (1997). LWRRDC Final report: Development and implementation of QA/QC protocols for sample processing components of the MRHI agency bioassessment program. Attachment 1. External QA/QC of MRHI subsampling and sorting proceedures. Internal Report 299, Supervising Scientist, Canberra and Darwin, Unpublished Report.

Humphrey, C.L., Thurtell, L., Pidgeon, R.W.J., van Dam R.A. and Finlayson, C.M. (1999). A model for assessing the health of Kakadu's streams. *Australian Biologist* **12**: 33-42.

Hynes, H.B.N. (1970). *The Ecology of Running waters*. Liverpool University Press, Liverpool.

Jackson, D.A. (1993). Multivariate analysis of benthic invertebrate communities: the implication of choosing particular data standardisations, measures of association, and ordination methods. *Hydrobiologia* **268**: 9-26.

Kaenel BR., Matthaei C.D. and Uehlinger, U. (1998). Disturbance by aquatic plant management in streams: Effects on benthic invertebrates. *Regulated Rivers-Research & Management* **14**: 341-356.

Karr, J.R. (1999). Defining and measuring river health, *Freshwater Biology* **41**: 221-234.

Kenkel, N.C and Orloci, L. (1986). Applying metric and non-metric multi-dimensional scaling to ecological studies: Some new results. *Ecology* **67**: 919-928.

Keough, M.J. and Mapstone, B.D. (1995). Protocols for Designing Marine Ecological Monitoring Programs Associated with BEK Mills. *National Pulp Mills Research Program Technical Report, CSIRO, Canberra* (No 11).

Kinhill and ERAES. (1996). *The Jabiluka Project: Main report: Draft Environmental Impact Statement.* Kinhill Engineers Pty Ltd in association with ERAES. Australia.

Koehn, J. and O'Connor, W. (1990). Threats to Victorian native freshwater fish, *Victorian Naturalist* **107**: 5-12.

Lemly, A.D (1982). Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. *Hydrobiologia* **87**: 229-245.

Lloyd. D.S. (1987). Turbidity as a water quality standard for salmonid habitats in Alaska. *North American Journal of Fisheries Management* **7**: 34-45.

Lloyd, D.S., Koenings, J.P., LaPerriere, J.D. (1987). Effects of turbidity in fresh waters of Alaska. *North American Journal of Fisheries Management* **7**: 18-33.

Mackay, R.J. (1992). Colonization by lotic macroinvertebrates: a review of processes and patterns. *Canadian Journal of Fisheries and Aquatic Sciences* **49**: 617-628.

Marchant. R. (1982a). Life spans of two species of tropical mayfly nymph (Ephemeroptera) from Magela Creek, Northern territory. *Australian Journal of Marine and Freshwater Research* **33**: 173-179.

Marchant. R. (1982b). Seasonal variation in the macroinvertebrate fauna of billabongs along Magela creek, Northern territory. *Australian Journal of Marine and Freshwater Research* **33**: 329-342.

Marchant, R. (1986). Some quantitative aspects of the life history of aquatic insects in temperate Australian rivers. *In* de Decker, P. and Williams. W.D. (eds). *Limnology in Australia.* CSIRO Melbourne and W.Junk. Dordrecht. 151-158.

Marchant. R. (1990). Robustness of classification and ordination techniques applied to macroinvertebrate communities from the La Trobe River, Victoria. *Australian Journal of Marine and Freshwater Research* **41**: 493-504.

Matthaei. C.D., Uehlinger, U., Meyer. E.I. and Frutiger, A. (1996). Recolonisation by benthic invertebrates after experimental disturbance in a Swiss prealpine river. *Freshwater Biology* **35**: 233-248.

Matthaei, C.D., Uehlinger, U. and Frutiger, A. (1997). Response of benthic invertebrates to natural versus experimental disturbance in a Swiss prealpine river. *Freshwater Biology* **37**: 61-77.

Matthaei, C.D., Peacock, K.A. and Townsend, C.R. (1999). Scour and fill patterns in a New Zealand stream and potential implications for invertebrate refugia. *Freshwater Biology* **42**: 41-57.

McCabe, Jr. G.T., Hinton, S.A and Emmett, R.L. (1998). Benthic invertebrates and sediment characteristics in a shallow navigation channel of the lower Columbia river, before and after dredging. *Northwest Science* **72**: 116-126.

Metzeling, L., Graesser, A., Suter, P., and Marchant, R. (1984). The distribution of aquatic macroinvertebrates in the upper catchment of the La Trobe River, Victoria. *Occasional Papers from the Museum of Victoria* **1**: 1-62.

Metzeling, L., Doeg, T., and OÇonnor, W. (1995). The impact of salinisation and sedimentation on aquatic biota. *In* Bradstock, R.A., Auld, T.D., Kieth, D.A., Kingston, R.T., Lunney, D., and Siversten, D.P. (eds). *Conserving Biodiversity: Threats and Solutions*. Surrey Beatty and Sons, London.

Mitchell, A.W., Bramley, R.G.V. and Johnson, A.K.L. (1997). Export of nutrients and suspended sediment during a cyclone-mediated flood event in the Herbert River catchment, Australia. *Marine and Freshwater Research* **48**: 79-88.

Newbold, J.D., Erman, D.C., and Roby, K.B. (1980). Effects of logging in streams with and without buffer strips. *Canadian Journal of Fisheries and Aquatic Sciences* **37**: 1076-1085.

Newcombe, C.P. and McDonald, D.D. (1991). Effects of suspended sediment on aquatic ecosystems. *North American Journal of Fisheries Management* **11**: 72-82.

Niemi, G.J., DeVore, P., Detenbeck, N., Taylor, D., Lima, A., Pastor, J., Yount, J.D. and Naiman, R.J. (1990). Overview of case studies on recovery of aquatic systems from disturbance. *Environmental Management* **14**: 571-587.

Norris, R.H. and Norris, K.R. (1995). The need for biological assessment of water quality: Australian perspective. *Australian Journal of Ecology* **20**: 1-6.

O'Connor, R., Humphrey, C., Dostine, P., Lynch, C. and Spiers, A. (1995). A survey of aquatic macroinvertebrates in lentic waterbodies of Magela and Nourlangie Creek catchments, Alligator Rivers Region, Northern Territory. *Internal Report 225, Supervising Scientist for the Alligator Rivers region, Canberra, Unpublished paper.*

O'Connor, R., Humphrey, C. and Lynch, C. (1997). Macroinvertebrate community structure in Magela Creek between 1988 and 1996: Preliminary analysis of monitoring data. *Internal report 261. Supervising Scientist for the Alligator Rivers region*, Canberra, Unpublished paper

Osenburg, C.W., Schmitt, R.J., Holbrook, S.J., Abu-Saba, K.E., and Flegel, A.R. (1994). Detection of environmental impacts: natural variability, effect size, and power analysis. *Ecological Applications* 4: 16-30.

OSS. (1993). *Office of the Supervising Scientist for the Alligator Rivers Region, Annual Report 1992-93*. AGPS, Canberra.

Outridge, P.M. (1988). Seasonal and spatial variations in benthic, macroinvertebrate communities of Magela Creek, Northern Territory. *Australian Journal of Marine and Freshwater Research* **39**: 211-223.

Paltridge, R. (1992). *Macroinvertebrate recolonisation of a tropical temporary stream. Honours Thesis*, Northern Territory University, Darwin.

Paltridge, R. M., Dostine, P.L., Humphrey, C.L. and Boulton, A.J. (1997). Macroinvertebrate recolonisation after re-wetting of a tropical seasonally flowing stream (Magela Creek, Northern Territory, Australia). *Marine and Freshwater Research* **48**: 633-645.

Peters, J.C (1967). Effects on a trout stream of sediment from agricultural practises, *Journal of Wildlife Management* **31**: 805-812.

Pickett, S.T.A. and White, P.S. (1985). *The Ecology of Natural disturbances and Patch Dynamics*. Academic Press, New York.

Poff, N.L. and Ward, J.V. (1989). Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. *Canadian Journal of Fisheries and Aquatic Sciences* **46**: 1805-1818.

Poff, N.L. (1992) Why disturbance can be predictable: a perspective on the definition of distrubance in streams. *Journal of the North American Benthological Society* **11**: 86-92.

Popperl, R. (1996). The structure of a macroinvertebrate community in a northern German lake outlet (Lake Belau, Schleswig-Holstein) with special emphasis on abundance, biomass and secondary production, *Internationale Rev. der Gesamten Hydrobiologie* :183-198.

Press, T. and Lawrence, D. (1995). Kakadu National Park: reconciling competing interests. *In* Press, T., Lea, D., Webb, A. and Graham, A. (eds). *Kakadu: Natural and Cultural Heritage and Management*. Australian Nature Conservation Agency and Australian National University, Canberra.

Quinn, J.M., Davies-Colley, R.J., Hickey, C.W., Vickers, M.L., Ryan, P.A (1992). Effects of clay discharges on streams. 2. Benthic invertebrates. *Hydrobiologia* **248**: 235-247.

Resh, V.H., Brown, A.V., Covich, A.P., Gurtz, M.E., Li , H.W., Minshall, G.W., Reice, S.R., Sheldon, A.L., Wallace, J.B and Wissmar, R. (1988). The role of disturbance in stream ecology. *Journal of the North American Benthological Society* **7**: 433-455.

Resh, V.H. and McElravy, D. (1993). Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. *In* Rosenburg, D.M. and Resh, V.H. (eds). *Freshwater Biomonitoring and Benthic Macroinvertebrates*, Chapman and Hall, New York.

Reynolds, J.B., Simmons, R.C., and Burkholder, A.R. (1989). Effects of placer mining discharge on health and food of Arctic Grayling. *Water Resources Bulletin* **25**: 625-635.

Richards, C., and Bacon, K.L. (1994). Influence of fine sediment on macroinvertebrate colonization of surface and hyporheic substrates. *Great Basin Naturalist* **54**: 106-113.

Richardson, B.A. (1985). The impact of forest road construction on the benthic invertebrate fauna of a coastal stream in southern New South Wales. *Australian Society for Limnology Bulletin* **10**, 65-88.

Rosenburg, D.M. and Resh, V.H. (1993). Introduction to freshwater biomonitoring and benthic macroinvertebrates. *In* Rosenburg, D.M. and Resh, V.H. (eds) *Freshwater Biomonitoring and Benthic Macroinvertebrates*, Chapman and Hall, New York.

Russell-Smith J, Needham S. and Brock J. (1995). The Physical Environment. *In* Press, T., Lea, D., Webb, A. and Graham, A. (eds). *Kakadu: Natural and Cultural Heritage and Management.* Australian Nature Conservation Agency and Australian National University, Darwin.

Ryan, P.A. (1991). Environmental effects of sediment on New Zealand streams: a review. *New Zealand Journal of Marine and Freshwater Research* **25**: 207-221.

Sainty, G.R. and Jacobs, S.W.L. (1994). *Waterplants in Australia*. Sainty and Associates, Sydney

Shaver, M.L., Shannon, J.P., Wilson, K.P., Benenati, P.L. and Blinn, D.W. (1997). Effects of suspended sediment and desiccation on the benthic tailwater community in the Colorado River, USA. *Hydrobiologia* **357**: 63-72.

Silsbee, D.G. and Larson, G.L. (1983). A comparison of streams in logged and unlogged areas of Great Smokey Mountains National Park. *Hydrobiologia* **102** 99-111.

Smith, R.W, Bernstein, B.B. and Climburg, R.L (1988). Community-environmental relationships in the benthos: applications of multivariate analytical techniques. *In* Soule, D.F and Kleppel, G.S (eds). *Marine Organisms as Indicators*, Springer Verlag, New York, 247-326.

Smith, E.P., Orvos, D.R., and Cairns, J.Jnr. (1993). Impact assessment using the Before-After-Control-Impact (BACI) model: concerns and comments. *Canadian Journal of Fisheries and Aquatic Sciences* **50**: 627-637.

Snedecor. G.W. and Cochran. W.G. (1989). *Statistical Methods, 8th edn.* Iowa State University Press, Iowa.

Statistica. (1995). *Volume 1: General Conventions and Statistica I.* Microsoft Corporation, USA

Stewart-Oaten, A., Murdoch, W.M., and Parker, K.R (1986). Environmental impact assessment: "psuedoreplication" in time? *Ecology* **67**: 929-940.

Stewart-Oaten, A., Bence, J.R., and Osenberg, C.W. (1992). Assessing the affects of unreplicated perturbations: no simple solutions. *Ecology* **73**: 1396-1404.

Stowar, M. (1997). Effects of suspended solids on the benthic macroinvertebrate fauna downstream of a road crossing, Jim Jim Creek, Kakadu National Park. *Supervising Scientist Internal Report* 256.

Thrush, S.F., Pridmore, R.D., and Hewitt, J.E (1994). Impacts on soft-sediment macro-fauna: the effects of spatial variation on temporal trends. *Ecological Applications* **4**: 31-41.

Townsend, S. (1997). The impact of fire management on stream water quality in the wet/dry tropics of northern Australia. *In* McKaige, B.J., Williams, B.J. and Waggitt, R.J (Eds). Proceedings of the Australian Bushfire Conference, Darwin, 8-10 July.

Townsend, S.A. and Douglas, M.M. (1997). The impact of fire management on the suspended solid concentrations of two tropical savanna streams in Australia. *Verhandlungen Internationale Vereinigung Limnologie* **26**: 887-889.

Townsend, C. R., Scarsbrook, M.R. and Doledec, S. (1997). Quantifying disturbance in streams: alternative measures of distrubance in relation to macroinvertebrate species traits and species richness. *Journal of the North American Benthological Society* **16**: 531-544.

Townsend, S.A. and Douglas, M.M. (2000). The effect of three fire regimes on stream water quality, water yield and export coefficients in a tropical savanna (northern Australia). *Journal of Hydrology* **229**: 118-137.

Tripodi, E. (1996). Temporal and spatial variations in the macroinvertebrate communities of the seasonally flowing portions of Magela Creek, Northern Territory. *Supervising Scientist Internal Report* 240.

Tumwesigye, C., Yusuf, S.K. and Makanga, B. (2000). Structure and composition of benthic macroinvertebrates of a tropical forest stream, River, Nyamwera, western Uganda. *Journal of African Ecology* **38**: 72-77.

Underwood, A.J. (1991). Beyond BACI: experimental designs for detecting human environmental impacts on temporal variations in natural populations. *Australian Journal of Marine and Freshwater Research* **42**: 569-587.

Underwood, A.J. (1993). The mechanics of spatially replicated sampling programs to detect environmental impacts in a variable world. *Australian Journal of Ecology* **18**: 99-116.

Underwood, A.J. (1994). On beyond BACI: sampling designs that might reliably detect environmental differences. *Ecological Applications* **4**: 3-15.

Underwood, A.J. (1997). *Experiments in Ecology: Their logical design and interpretation using analysis of variance.* Cambridge University Press, Cambridge.

Walsh, C.J. (1997). A multivariate method for determining optimal subsample size in the analysis of macroinvertebrate samples. *Marine & Freshwater Research* **48**: 241-248.

West, G., Blyth, J.D., Balkau, F., and Slater, S. 1984. Chemical and physical features of construction sediment in the Mitta Mitta River, Victoria. *Occasional Papers of the Museum of Victoria* **1**: 129-138.

Wellings, P. (1995). Management Considerations. *In* Press, T., Lea, D., Webb, A. and Graham, A. (eds). *Kakadu: Natural and Cultural Heritage and Management.* Australian Nature Conservation Agency and Australian National University, Canberra.

White, D.S. and Gammon, J.R. (1977). The effect of suspended solids on macroinvertebrate drift in an Indiana Creek. *Proceedings Indiana Academy of Sciences* **86**: 182-188.

Williams, W.D. (1980). *Australian Freshwater Life: The Invertebrates of Australian Inland Waters* 2nd edition. Macmillan Education Australia, Pty Ltd, Hong Kong.

Williams, D.D. and Hynes, H.B.N. (1976). The recolonisation mechanisms of stream benthos. *Oikos* **27**: 265-272.

Williams, D.D and Williams, N.E. (1993). The upstream/downstream movement paradox of lotic invertebrates: quantitative evidence from a Welsh mountain stream. *Freshwater Biology* **30**: 199-218.

Wright, I.A., Chessman, B.C., Fairweather, P.G. and Benson, L.J. (1995). Measuring the impact of sewage effluent on the macroinvertebrate community of an upland stream: The effect of different levels of taxanomic resolution and quantification. *Australian Journal of Ecology* **20**: 142-149.

Yount, J.D. and Niemi. G.J. (1990). Recovery of lotic communities and ecosystems from disturbance-A narrative review of case studies. *Environmental Management* **14**: 547-569.

Appendix 1: Review of studies on the effects of suspended sediment

Sediment, which either remains suspended in the water column or deposited out onto the substrate, becomes a problem when duration of exposure or concentrations exceed levels normally found in the natural systems (Newcombe and MacDonald 1991; Hellawell 1986). If the sediment remains in suspension, it can cause a decrease in water clarity and add to the non-nutritional status of material in suspension for filter-feeders, or it can physically harm aquatic biota (Hellawell 1986). If sediment settles onto the substrate, it can coat and scour the surface, fill in vital interstitial pores that provide habitats for invertebrates, or reduce periphyton (Hellawell 1986; Richards and Bacon 1994).

Forestry Activities

Forestry activities such as clear-felling result in the release of large quantities of organic matter previously bound up in the soil and tree root systems. This organic enrichment not only creates increased sediment loads but can create anoxic conditions for aquatic flora and fauna (Ryan 1991). The results of several studies highlight how sites downstream of logging activities show a decrease in diversity and/or abundance, of benthic invertebrate communities (Newbold et al 1980; Growns and Davis, Davies and Nelson 1994;). Davies and Nelson (1994) examined the effects of logging a Tasmanian forest over several seasons and found changes in the taxa of downstream receiving waters. Most reduced in abundance were stoneflies, Leptophlebiidae mayflies and the Brown trout. Growns and Davis (1994) found a decrease in both abundance and diversity of the downstream taxa at four treatment sites in a south-west Western Australian forest after four months of clear-felling, however, recovery of the fauna was rapid when sediment loads returned to pre-impact levels.

Another response to disturbance can be the increase in numbers of opportunistic taxa which benefit from disrupted conditions through having rapid colonisation rates or short generation times (Hynes 1970). Newbold et al (1980) found the opportunists, Chironomidae and Baetidae (*Baetis*), occurred in large numbers at logged sites, thereby reducing overall community diversity. Some Chironomidae species have the ability to burrow into the hyporrheic zone where a more favourable oxygen environment is available (Gunn 1997). A 3-fold increase in Chironomidae was found one year after logging, combined with a decrease in Ephemeroptera, Odonates and Plecoptera taxa for the Piscataquis River in the U.S.A after widespread deforestation in the catchment (Garman and Moring 1993). Silsbee and Larson (1983) found a pattern of increased abundance in general in logged streams 40 years on from initial logging practices. They postulated that this pattern might be due to an increase of allochthonous material from secondary regrowth.

Construction Activities - Roads and Dams

Road construction studies undertaken in the 1970s and early 1980's showed changes to species composition resulted from increased suspended solid loads entering streams (Barton 1977; Extence 1978; Cline et al 1982). The filter feeding Hydropsychidae showed reduced density in Extence's (1978) study, as did other taxa which rely on a stable substratum. Barton (1977) and Cline et al (1982) both recorded high maximum levels of suspended sediment; 70-500 mg/L (<10 mg/L background) over 4 months, and 1390 mg/L (<5 mg/L background) over 8 months respectively. Stowar (1997) looked at macroinvertebrate communities downstream of a road crossing in Kakadu National Park and found that maximum turbidity levels of 60 NTU's, corresponding to suspended solid loads of 100mg/L, against a baseline level of >5 NTU's, over a 3 month period, were sufficient to cause the loss of Chironomidae from the downstream community.

Studies of both Dartmouth and Thompson Dams, in eastern Australia, have shown that construction activities increase the loads of suspended sediment to streams (West et al 1984) leading to deposition far downstream, in some cases up to 33km (Davey et al 1987). Algae are able to profit from increased nutrient loads that adsorb to suspended particulate matter; when flow rates are suitable, algae proliferate. The appearance of a layer of algae on rocks in riffle areas downstream of the Dartmouth dam caused severe changes to benthic macroinvertebrate communities. Community structure became dominated by a few tolerant taxa such as Oligochaeta and some Chironomids (West et al 1984; Blyth et al 1984). Doeg et al (1987) found an effect 33km below the Thompson dam construction site where total abundance and diversity of samples were all lower than upstream locations. Chessman et al (1987) found that while improved erosion control reduced the impact from construction at Blue Rock Dam on the Tanjil River, faunal density was still reduced by 30-40% at downstream locations. Turbidity levels of 480 NTU were reached, however the range as a pulse effect over 3 years was between 1-110 NTU (Chessman et al 1987).

Sediment Addition from Reservoirs, Dredging and other Land Use Practices

Hogg and Norris (1991) studied the effects of urban land clearing on the Murrumbidgee River and found reduced numbers of invertebrates in the downstream pools with Oligochaetes dominating these sites. This was attributed to alterations to the substrate by the deposition of fine inorganic particulates. Levels of up to 560 mg/L were reached above a background of <5mg/L as brief pulses over several years preceding the study. In contrast, McCabe et al (1998) and Harvey (1986) both noted no statistically significant changes to the invertebrates sampled at sites affected by dredging activities; the substrate did not appear to be sufficiently altered in either study to have deleterious effects. Harvey (1986) reported that maximum levels of 50 NTU were reached but that these levels were short-lived events.

Recovery from disturbance can be very rapid when the sediment remains in suspension because it is washed through the system more quickly. The flushing of sediment from a reservoir in Wyoming caused the suspension of fine particulate matter with no deposition occurring downstream. A 20-fold increase in suspended sediment was recorded over a two month period. As a result, chironomid populations decreased by 90% during this time with Oligochaetes increasing in abundance, however, a return to pre-impact abundances for these taxa took only 3 weeks (Gray and Ward 1982).

Placer (alluvial) gold mining provides another set of studies where the pollutant is almost exclusively suspended sediments. Research in New Zealand has shown that clay particles in the water column caused an increase in turbidity from 2.4 NTU upstream to >100 NTU (median 15 NTU) downstream of mining (Davies-Colley et al 1992). A study undertaken 2-8 years post-disturbance revealed levels of 154 NTU against a background of <8 NTU, resulting in significant reductions in invertebrate densities and taxonomic richness, especially where turbidity loads remained high (Quinn et al 1992). A lowered productivity resulting from the degraded food supply was considered the main reasons for these effects.

Appendix 2 Review of adverse effects on aquatic biota from suspended sediment

Effects on macroinvertebrates

When sediment settles onto the substrate it can cover the periphyton which is a major food supply for most grazing benthic taxa thereby reducing abundance through scouring and light reduction for photosynthesis (Ryan 1991; Newcombe and McDonald 1991). The filter feeding taxa are affected through clogging of their gills which results in stress, growth reduction and also death (Newcombe and McDonald 1991). Filter feeders are also susceptible to starvation when too much inert matter dominates in their food supply thereby decreasing the nutritional value of the diet (Hellawell 1986). Respiratory processes can also be affected by fine silt particles causing oxygen deprivation in the organism (Chutter 1968).

Substrate affects macroinvertebrate assemblage structure directly through creating different sized interstitial spaces for colonisation (Richards and Bacon 1994). Fine sediment deposition can fill these interstitial spaces thus reducing available habitat for hyporheic taxa. Sediment deposition onto the substrate can influence the assemblage structure as certain taxa require smooth surfaces for colonisation e.g. Simuliidae (Chutter 1968). Stability of the substrate can also be affected if stable habitat is replaced by less stable particles making it difficult for certain taxa to maintain an attachment (Hellawell 1986).

Drift is a phenomenon of movement of benthos in the water column that can be either part of normal behaviour (i.e. diurnal drift) or a result of disturbance. Catastrophic drift, as the latter is termed, is a behavioural response to contamination, including increased suspended sediment, where the benthos move away from the location of the pollutant in high numbers, resulting in altered community structure (Fairchild et al 1987; Doeg and Milledge 1991).

The general result of sediment pollution is to alter community structure i.e. diversity and abundance, with the loss of species intolerant to altered conditions and replacement by more tolerant forms such as the burrowers, e.g. Oligochaetes and Chironomid larvae (Hellawell 1986). This change in community structure has a cascading effect to higher trophic levels.

Effects on fish

Fish respond to increased suspended solid loads either directly or indirectly through trophic changes to the secondary production, such as to their food supply in the benthos. Effects can be either lethal or sub-lethal depending on the level of concentration of sediment. Direct effects can stem from reducing respiratory and feeding efficiency with the added burden of silt particles in the water column (Hellawell 1986). Respiration becomes affected as a result of increased mucus secretion by the gills. As food is filtered through the mouth and gills suspended particles can cause irritation resulting in increased mucus secretions to off-set this effect. The mucus attracts sediment particles that together form a layer that can inhibit respiration (Reynolds et al 1989). This can also lead to reduced growth rates and increased stress amongst the population (Lloyd 1987).

Turbid waters make sighting prey, as for the Arctic grayling and salmon, much more difficult (Reynolds et al 1989). This can result in avoidance behaviour where clearer waters are sought out (Bisson and Bilby 1982). As a result, feeding rates may be affected as planktivorous fish, which are selective of prey size switch to feed upon the larger prey classes. This behaviour also affects the zooplankton community structure (Gardner 1981). Visual disturbance has been found to interfere with cues for migration patterns and disorientate fish in this manner (Lloyd 1987).

Fish, together with macroinvertebrates, rely upon the substratum for stages of their life cycles. Interstitial pores provide spaces for egg-laying. Fish such as the Macquarie Perch lay their eggs amongst gravel substrate and this can be smothered by deposited sediment creating anoxic conditions for egg development (Peters 1967). Other fish species lay adhesive eggs and suitable sites need to be relatively free from silt and attached particles (Koehn and O'Connor 1990). Reynolds et al (1989) measured turbidity levels as high as 1000 NTU over a study period and found that eggs (sac fry) were extremely vulnerable at these high levels through the aforementioned processes.

Effects on aquatic plants

Light penetration is affected by increased particles of suspended sediment in the water column and this can adversely effect aquatic vegetation dependant on this energy source for growth (Hellawell 1986). Growth rates and biomass of periphyton can be affected by the nutrients that adsorb onto sediment particulate matter (Newcombe and MacDonald 1991). Turbidity levels as low as 5 NTU can cause a subsequent decrease in primary productivity by as much as 3-13% and an increase of 25 NTUs may cause a reduction of between 13-50% in primary productivity (Lloyd 1987). Van Nieuwenhuyse and LaPerriere (1986, cited in Ryan 1991) measured primary productivity in mined and unmined streams in Alaska. They discovered that turbidity levels of 170 NTU caused primary productivity to be reduced by 50%. A complete loss of primary productivity occurred at 1200 NTU.

Experimental Studies

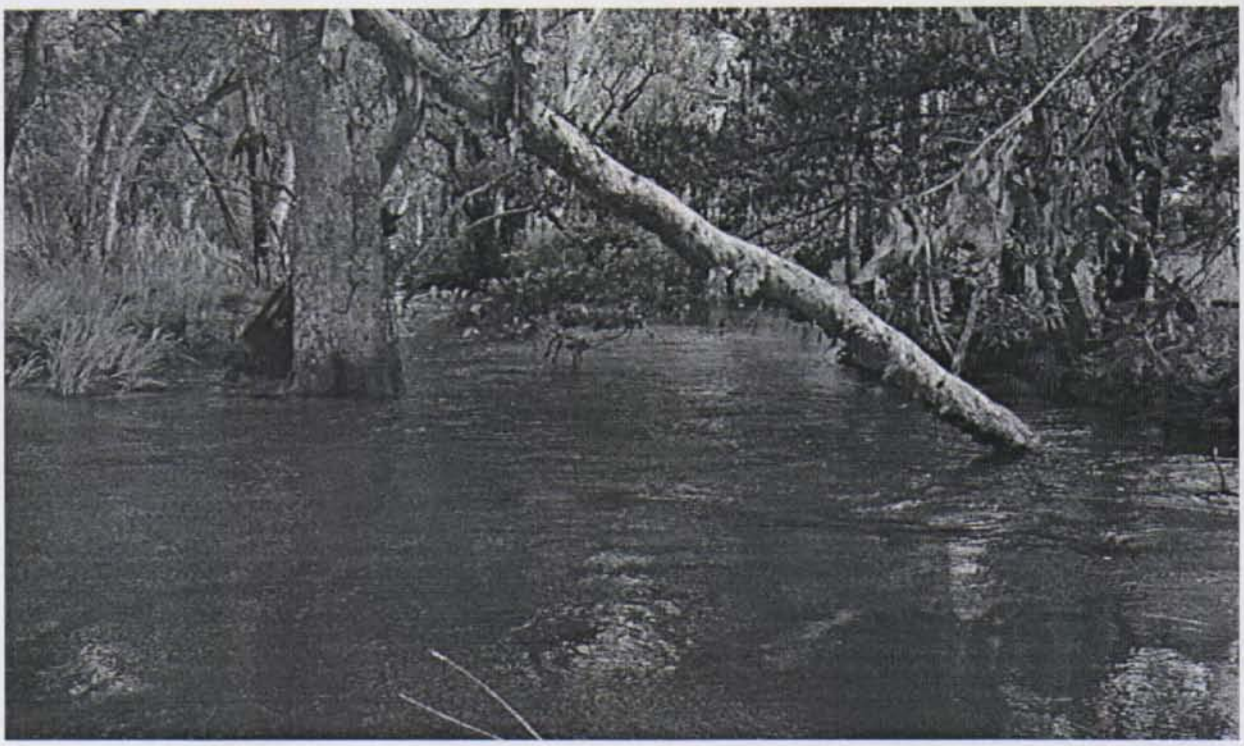
Several studies have attempted to determine the level at which suspended sediment constitutes a pollutant in waterways. These types of experiments are still only relevant to their particular local conditions of environment, climate and disturbance. As with most experimental studies, it is difficult to extrapolate with any certainty the conclusions to other situations.

Doeg and Milledge (1991) experimentally altered sediment input levels over 1.5 hours in a channel in north-eastern Victoria. Levels were altered every 15 minutes and it was found that the level of 133mg/L (suspended solids) initiated catastrophic drift by benthic invertebrates. This concentration threshold resulted in a 7-fold increase in total abundance of invertebrates in the drift. Several taxa appeared to be more sensitive to the sediment addition. Fairchild et al (1987) carried out a similar experiment where sediment at levels of 1700 mg/L was added to a stream every 2 hours over a six week period. This increase in loads of sediment altered drift dynamics of the invertebrates by delaying nocturnal drift patterns. Net nutrient retention increased as well. This level of sediment addition was considered to be low yet drift dynamics appeared to be sensitive even at this level. Community structure was unaffected by the sediment. Culp et al (1986) added fine sediment to a stream in order to assess the effects of suspension and deposition upon benthic communities. It was found that transport of sediment (saltation) caused catastrophic drift, reducing benthic density by 50% within 24 hours. The deposition of sediment did not have any noticeable effect upon stream biota apart from one taxon.

Experiments have also investigated trophic changes to macroinvertebrate communities by the addition of suspended sediment. Shaver et al (1997) examined two algal communities under different suspended sediment loads, one turbid the other not. They found that higher levels of sediment caused proliferation of one algae over another thereby reducing macroinvertebrate biomass. The algae *Oscillatoria*, coped well with sedimented conditions, and supported significantly less macroinvertebrates. Bunn (1987) found that an important pathway of food processing was interrupted by sedimentation. Lotic systems rely upon the spiralling of nutrients and a reduction in density of shredders will in turn upset benthic community balance.

Appendix 6: photos of sampling sites

Suspended sediment entering Swift Creek Downstream 1 from the North tributary during early wet season flows in January, 1999



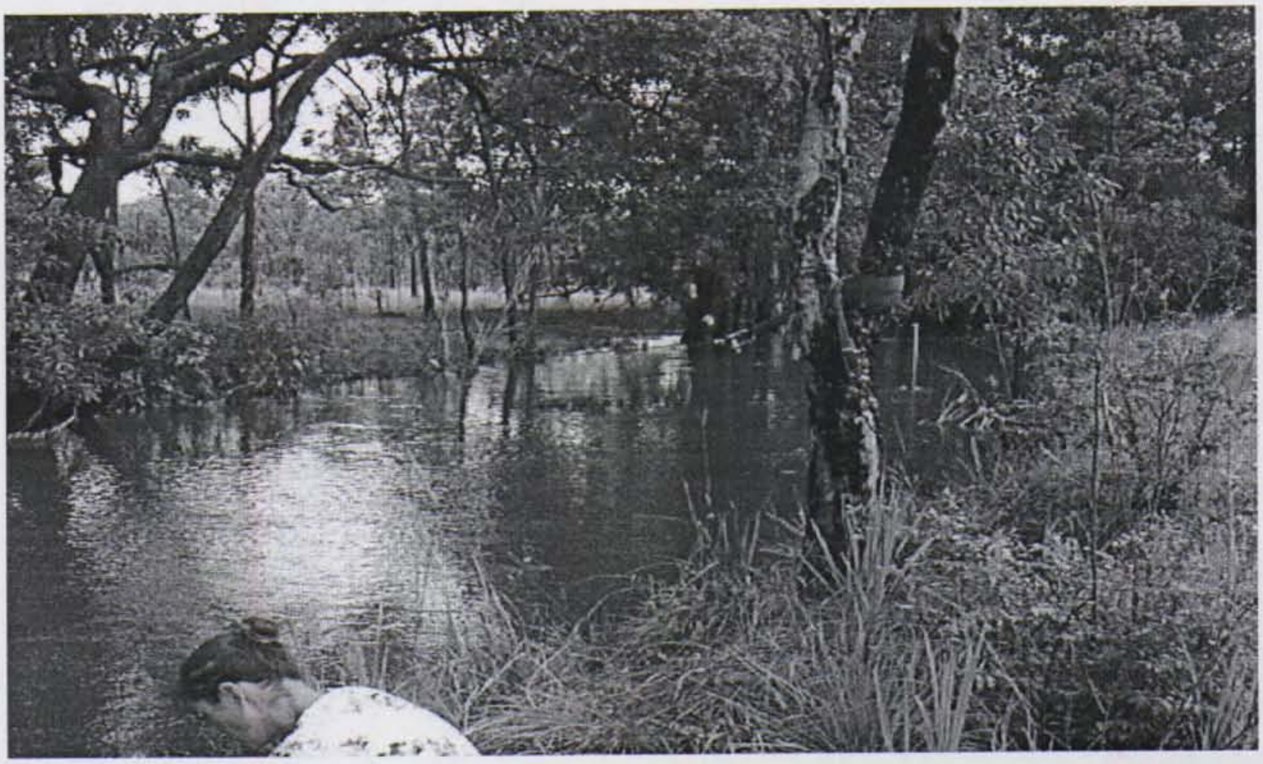
Swift Creek Downstream 2

Swift Creek Tributary West Upstream

Swift Creek Upstream Main

North tributary flowing past the Jabiluka Mine in late April, 1999

Suspended sediment entering Swift Creek from North Tributary during early wet season flows in January, 1999



7J Downstream

7J Upstream

North Magela Downstream

Catfish Creek Downstream

Appendix 4: Methods used in Water Chemistry Analysis

Variable	Method	Reference
pH	Electrometric	APHA*2510D
Alkalinity	Acidimetric titration	APHA*2320B
Conductivity	Electrometric	APHA*2510D
Turbidity	Nephelometric	APHA*2130B
Orthophosphate	Spectrophotometric	APHA*4500P E
Total phosphorous	Acid digestion and spectrophotomeric	APHA*4500P E
Total organic Carbon/ Dissolved Organic Carbon	Acidifcation, persulfate oxidation	APHA*5310D
Na+, K+, NH4-	HPLC	ERISS*
$S0_4^{2-}$, Cl-, $N0_3$-, Ca^{2+}, Mg^{2+}	HPLC	ERISS*
Al, Mn, Fe	ICP-AES	External*
Cu, Pb, Zn, U, Ni, Cd, Cr	ICP-MS	External*

*APHA-as specified in APHA (1995) Standard methods for the Examination of Water and Wastewater, 19th Edition, American Public Health Association, Washington.

*ERISS- unpublished method developed on site at the analytical chemistry laboratory in Jabiru, NT.

*External-analysed by ChemNorth Laboratory at Berrimah, Darwin, NT.

Appendix 5: General water chemistry analysis for main sampling sites

Variable	Sites	Sampling Dates						
		21-23/12/98	12-14/1/99	2-4/2/99	22-24/2/99	15-22/3/99	19-23/4/99	10-12/5/99
pH	STW	6.02	5.60	5.21	5.52	5.21	5.68	6.27
	SD1	4.82	5.08	4.84	5.16	5.12	5.25	5.75
	SD2	4.52	5.10	4.50	4.93	4.90	5.47	5.72
	SUM	NR	4.47	4.12	3.97	4.43	5.11	5.41
	NMU	5.89	6.09	5.60	6.03	5.83	5.45	6.39
	NMD	6.00	6.22	5.70	6.27	6.18	6.26	6.27
	7JU	4.87	4.67	5.67	5.06	5.01	5.87	5.85
	7JD	4.80	5.37	5.34	5.50	5.63	5.92	6.14
	CFU	4.40	4.90	4.28	5.00	4.08	4.16	4.79
	CFD	4.52	4.91	4.27	4.76	4.12	4.07	4.93
Alkalinity (mg/L $CaCo_3$)	STW	3.48	1.52	1.24	NR	0.48	3.10	NR
	SD1	0.10	0.20	0.36	NR	0.42	0.64	NR
	SD2	0.10	0.26	0.36	NR	0.56	0.94	NR
	SUM	NR	0.10	0.00	NR	0.10	0.10	NR
	NMU	2.70	3.66	3.84	NR	3.94	NR	NR
	NMD	2.84	3.52	4.16	NR	3.76	NR	NR
	7JU	0.28	0.18	0.34	NR	0.56	NR	NR
	7JD	0.10	0.82	2.04	NR	1.28	NR	NR
	CFU	0.10	0.10	0.72	0.10	0.10	NR	NR
	CFD	0.10	0.10	0.00	0.10	0.10	NR	NR
EC (mS/cm)	STW	20.0	10.0	9.1	9.0	5.0	11.0	18.5
	SD1	12.0	10.0	11.0	10.0	6.0	9.0	13.0
	SD2	NR	10.0	11.6	9.0	6.0	9.0	14.0
	SUM	NR	13.0	13.5	11.0	7.0	11.5	NR
	NMU	11.0	16.0	17.0	14.0	12.0	NR	16.0
	NMD	13.0	16.0	17.5	14.0	12.0	14.0	16.0
	7JU	9.0	10.0	8.2	8.0	7.0	11.5	14.5
	7JD	9.0	10.0	10.0	9.0	7.0	7.0	15.0
	CFU	13.5	14.2	9.8	8.0	7.9	9.0	15.5
	CFD	14.0	12.0	10.3	8.0	8.7	10.6	14.0
Turbidity (NTU)	STW	23.70	23.00	3.39	1.00	7.00	2.00	1.60
	SD1	44.00	4.43	2.74	2.00	5.00	1.00	0.75
	SD2	2.35	2.00	3.30	3.00	6.00	2.00	2.20
	SUM	NR	1.00	1.05	1.00	6.00	0.63	0.40
	NMU	113.00	16.00	5.70	6.00	7.00	0.05	2.00
	NMD	48.00	13.00	5.30	7.00	6.00	3.00	1.90
	7JU	9.60	1.00	11.91	3.00	4.00	NR	1.50
	7JD	12.00	2.00	17.94	3.00	4.00	4.00	1.53
	CFU	0.81	0.58	1.45	1.00	1.00	0.15	NR
	CFD	0.43	0.75	1.50	4.00	0.90	2.70	2.23
TOC (mg/L)	STW	7.34	3.83	2.20	2.03	2.55	NR	NR
	SD1	4.70	2.71	2.79	2.25	3.29	NR	NR
	SD2	5.37	2.79	2.86	2.34	3.57	NR	NR
	SUM	NR	2.85	2.76	2.14	3.09	NR	NR
	NMU	6.34	5.26	3.30	3.32	3.90	NR	NR
	NMD	6.23	4.94	3.30	3.25	3.82	NR	NR
	7JU	6.33	3.33	4.03	2.71	3.26	NR	NR
	7JD	6.74	3.55	3.54	2.91	3.16	NR	NR
	CFU	4.36	3.81	4.56	2.67	NR	NR	NR
	CFD	4.60	3.64	4.65	2.59	NR	NR	NR
DOC (mg/L)	STW	7.47	3.82	2.20	1.82	2.75	NR	NR
	SD1	4.72	2.90	2.70	1.99	3.31	NR	NR
	SD2	5.45	3.02	2.76	2.01	3.65	NR	NR
	SUM	NR	3.01	2.65	1.98	3.05	NR	NR
	NMU	6.07	5.42	3.32	2.94	3.47	NR	NR
	NMD	6.11	4.90	3.05	2.84	3.59	NR	NR
	7JU	6.73	3.51	3.95	2.48	2.99	NR	NR
	7JD	6.97	3.72	3.52	2.47	3.00	NR	NR
	CFU	4.28	3.84	4.29	2.54	NR	NR	NR
	CFD	4.63	3.75	3.97	2.35	NR	NR	NR

Appendix 6: Analysis of metals for main sampling sites

Variable	Site	Date of sampling						
		21-23/12/98	12-14/1/99	2-4/2/99	22-24/2/99	15-22/3/99	19-23/4/99	10-12/5/99
Al	STW	65	94	16	11	76	8	35
	SUM	NR	55	63	43	21	24	NR
	SCD1	65	48	46	32	57	17	34
	SCD2	80	48	51	30	61	17	53
	NMU	73	26	62	42	16	NR	14
	NMD	34	24	40	28	16	NR	13
	7JU	95	65	12	10	50	21	57
	7JD	104	99	13	9	38	14	36
	CFU	95	79	89	47	64	36	43
	CFD	105	71	96	43	64	42	49
Cd	STW	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	SUM	NR	0.00	0.00	0.00	0.00	0.00	NR
	SCD1	0.02	0.00	0.00	0.00	0.00	0.00	0.00
	SCD2	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	NMU	0.00	0.00	0.00	0.00	0.00	NR	0.00
	NMD	0.00	0.00	0.00	0.00	0.00	NR	0.00
	7JU	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	7JD	0.03	0.00	0.03	0.00	0.00	0.00	0.00
	CFU	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	CFD	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cr	STW	0.2	0.1	0.3	0.2	0.1	0.2	0.2
	SUM	NR	0.2	0.3	0.2	0.1	0.2	NR
	SCD1	0.1	0.2	0.4	0.2	0.1	0.2	0.2
	SCD2	0.2	0.2	0.3	0.2	0.1	0.2	0.2
	NMU	0.2	0.3	0.3	0.3	0.2	NR	0.1
	NMD	0.4	0.3	0.3	0.2	0.1	NR	0.2
	7JU	0.2	0.2	0.3	0.3	0.1	0.2	0.2
	7JD	0.2	0.3	0.3	0.2	0.1	0.2	0.2
	CFU	0.2	0.1	0.2	0.2	0.1	0.2	0.1
	CFD	0.2	0.1	0.2	0.2	0.1	0.2	0.1
Cu	STW	0.10	0.05	0.05	0.12	0.12	0.10	0.10
	SUM	NR	0.05	0.04	0.12	0.12	0.20	NR
	SCD1	0.08	0.03	0.11	0.14	0.14	0.14	0.07
	SCD2	0.11	0.02	0.08	0.13	0.13	0.15	0.24
	NMU	0.70	0.26	0.09	0.14	0.28	NR	0.13
	NMD	0.68	0.27	0.15	0.16	0.27	NR	0.13
	7JU	0.07	0.02	0.20	0.25	0.13	0.19	0.07
	7JD	0.10	0.07	0.26	0.25	0.13	0.16	0.08
	CFU	0.04	-0.03	0.01	0.11	0.20	-0.05	0.11
	CFD	0.08	-0.02	0.01	0.17	0.17	0.03	0.10
Fe	STW	137	220	46	57	82	41	451
	SUM	NR	102	74	67	48	43	NR
	SCD1	82	65	61	57	77	43	352
	SCD2	164	70	63	53	69	46	540
	NMU	136	105	72	66	67	NR	277
	NMD	109	101	59	55	67	NR	274
	7JU	96	122	62	70	72	40	492
	7JD	112	254	67	70	59	39	410
	CFU	74	47	62	49	95	39	78
	CFD	98	65	89	59	89	147	225
Mn	STW	8.05	4.42	2.01	1.78	2.71	2.59	3.48
	SUM	NR	3.76	3.50	2.58	1.81	2.85	NR
	SCD1	4.57	3.78	3.48	2.64	3.04	2.63	3.28
	SCD2	9.28	4.05	3.60	2.65	3.15	2.59	3.41
	NMU	9.20	5.37	4.59	3.04	4.11	NR	3.47
	NMD	6.57	4.50	4.02	2.71	4.31	NR	3.83
	7JU	5.12	3.83	3.95	4.05	3.16	2.76	4.85
	7JD	6.32	4.69	4.43	4.13	3.00	2.66	3.55
	CFU	3.78	2.25	2.31	1.71	1.82	1.61	2.89
	CFD	4.30	2.31	2.64	1.88	2.20	2.28	3.98

(All values in mg/L)

Appendix 6 continued

Variable	Site	Date of sampling						
		21-23/12/98	12-14/1/99	2-4/2/99	22-24/2/99	15-22/3/99	19-23/4/99	10-12/5/99
Ni	STW	0.13	0.11	0.10	0.08	0.06	0.10	0.16
	SUM	NR	0.06	0.06	0.07	0.06	0.13	NR
	SCD1	0.04	0.08	0.12	0.10	0.08	0.28	0.10
	SCD2	0.10	0.07	0.10	0.08	0.26	0.10	0.22
	NMU	0.17	0.22	0.11	0.10	0.24	NR	0.10
	NMD	0.20	0.15	0.13	0.16	0.14	NR	0.10
	7JU	0.07	0.06	0.15	0.15	0.09	0.21	0.11
	7JD	0.09	0.08	0.45	0.15	0.09	0.16	0.10
	CFU	0.14	0.00	0.03	0.04	0.08	0.04	0.02
	CFD	0.15	-0.01	0.04	0.06	0.18	0.07	0.03
Pb	STW	0.02	0.01	0.04	0.01	0.01	0.00	0.00
	SUM	NR	-0.01	-0.03	0.01	0.01	0.02	NR
	SCD1	0.04	-0.01	0.21	0.07	0.02	0.01	0.00
	SCD2	0.03	-0.03	0.14	0.01	0.01	0.00	0.02
	NMU	0.05	0.01	0.18	0.01	0.01	NR	0.00
	NMD	0.04	0.01	0.35	0.01	0.01	NR	0.00
	7JU	0.02	-0.01	0.10	0.03	0.01	0.03	0.00
	7JD	0.12	0.00	0.61	0.01	0.02	0.05	0.01
	CFU	0.01	-0.03	-0.05	0.01	0.00	0.00	0.00
	CFD	0.05	-0.05	-0.05	0.02	0.02	0.03	0.00
U	STW	0.038	0.035	0.018	0.012	0.009	0.013	0.023
	SUM	NR	0.010	0.010	0.007	0.013	0.004	NR
	SCD1	0.011	0.014	0.013	0.008	0.011	0.008	0.007
	SCD2	0.019	0.012	0.012	0.008	0.011	0.007	0.013
	NMU	0.045	0.055	0.023	0.011	0.020	NR	0.012
	NMD	0.044	0.033	0.024	0.011	0.020	NR	0.012
	7JU	0.018	0.013	0.025	0.019	0.011	0.011	0.013
	7JD	0.015	0.022	0.032	0.020	0.012	0.010	0.013
	CFU	0.008	0.006	0.009	0.005	0.005	0.003	0.003
	CFD	0.008	0.010	0.010	0.005	0.006	0.004	0.006

(All values in mg/L)

Appendix 7: Results from water chemistry analysis of the tributaries, Central, North and South tributaries, draining the mine site into Swift Creek

Site	Alkalinity	EC	pH	Turbidity
	(mg/L $CaC0_3$)	(uS/cm)		(NTU)
Central 1	NR	18.0	6.23	0.80
Central 2	NR	12.0	6.21	1.00
Central 3	NR	16.5	6.15	1.20
Central 4	NR	15.0	5.99	NR
Central 5	NR	15.0	6.03	NR
Central 6	NR	15.5	6.13	2.33
North 1	2.42	32.0	5.97	11.00
North 2	5.18	38.0	6.06	6.20
North 3	0	50.0	4.38	13.50
North 4	0.68	31.0	5.52	6.80
North 5	0	20.0	5.56	2.10
North 6	NR	19.5	5.67	2.20
South 2	NR	8.4	5.50	0.44
South 3	NR	9.6	5.34	0.40
South 4	NR	8.2	5.47	16.00
South 5	NR	8.6	5.34	0.63
South 6	NR	9.0	5.36	6.00

Appendix 8: Macroinvertebrate raw data

Site	7JD			7JU			CFD			CFU			NMD			NMU		
Sampling Date	22/12/98			22/12/978			23/12/98			23/12/98			22/12/98			22/12/98		
Replicate	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
Acarina	2.4	0.0	5.0	1.0	0.0	1.3	164.3	77.3	200.0	120.0	325.0	650.0	37.5	18.8	6.3	0.0	4.0	4.5
Aeshnidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0
Amphisopodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ancylidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Anisoptera Ind	0.0	15.8	2.5	0.0	0.0	1.3	7.1	0.0	20.0	0.0	75.0	300.0	75.0	6.3	12.5	0.0	4.0	0.0
Arachnid	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Atyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Baetidae	2.4	10.5	0.0	2.0	4.5	0.0	92.9	18.2	160.0	500.0	475.0	1050.0	62.5	56.3	18.8	4.8	8.0	4.5
Belostomatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Caenidae	7.1	5.3	2.5	5.0	4.5	12.5	100.0	31.8	420.0	260.0	325.0	700.0	175.0	31.3	18.8	2.4	20.0	27.3
Calamoceratidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ceratopogonidae L	2.4	26.3	0.0	1.0	4.5	1.3	42.9	4.5	160.0	20.0	0.0	0.0	25.0	37.5	50.0	11.9	4.0	9.1
Ceratopogonidae P	7.1	5.3	15.0	1.0	0.0	1.3	0.0	0.0	0.0	0.0	0.0	50.0	50.0	25.0	18.8	2.4	8.0	54.5
Chironomidae L	14.3	115.8	22.5	3.0	31.8	13.8	207.1	113.6	300.0	360.0	350.0	550.0	700.0	306.3	218.8	78.6	48.0	218.2
Chironomidae P	2.4	10.5	5.0	3.0	4.5	2.5	0.0	9.1	40.0	0.0	50.0	100.0	250.0	81.3	12.5	26.2	124.0	54.5
Chrysomelidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cladocera	14.3	0.0	2.5	2.0	0.0	1.3	142.9	109.1	780.0	640.0	500.0	1200.0	512.5	256.3	50.0	59.5	104.0	177.3
Coenagrionidae	0.0	0.0	0.0	0.0	0.0	0.0	14.3	4.5	3.0	20.0	0.0	0.0	37.5	0.0	0.0	0.0	0.0	0.0
Collembola	2.4	5.3	10.0	2.0	0.0	2.5	14.3	4.5	0.0	0.0	0.0	0.0	37.5	12.5	0.0	4.8	4.0	0.0
Copepod	4.8	26.3	0.0	0.0	0.0	1.3	50.0	18.2	100.0	180.0	50.0	250.0	112.5	6.3	50.0	7.1	8.0	50.0
Corbiculidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corduliidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.5	6.3	0.0	2.4	0.0	4.5
Corixidae	0.0	5.3	0.0	0.0	0.0	0.0	335.7	113.6	820.0	200.0	50.0	1350.0	0.0	31.3	18.8	0.0	4.0	0.0
Culicidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0
Culicidae L	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Culicidae P	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Decapod Ind	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dytiscidae A	0.0	10.5	2.5	0.0	4.5	0.0	14.3	9.1	20.0	120.0	25.0	100.0	12.5	0.0	0.0	0.0	0.0	0.0
Dytiscidae L	0.0	0.0	1.0	1.0	0.0	0.0	28.6	9.1	20.0	0.0	25.0	400.0	12.5	6.3	0.0	0.0	4.0	0.0
Ecnomidae	0.0	0.0	0.0	0.0	0.0	0.0	1.0	4.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.8	0.0	4.5
Elmidae A	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	20.0	0.0	25.0	50.0	0.0	0.0	0.0	0.0	0.0	0.0
Elmidae L	0.0	15.8	5.0	1.0	0.0	1.3	7.1	9.1	40.0	40.0	25.0	50.0	0.0	0.0	0.0	0.0	0.0	0.0
Ephemeroptera Ind.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gerridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gomphidae	0.0	10.5	0.0	0.0	1.0	1.3	0.0	0.0	1.0	40.0	0.0	100.0	12.5	1.0	0.0	1.0	4.0	1.0
Gyrinidae A	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydraenidae A	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydraenidae L	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrometridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae A	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydrophilidae L	0.0	0.0	0.0	0.0	4.5	0.0	14.3	0.0	1.0	80.0	50.0	250.0	25.0	18.8	0.0	0.0	0.0	0.0
Hydropsychidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydroptilidae A	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydroptilidae L	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	50.0	25.0	0.0	0.0	0.0	0.0	4.5
Hydroptilidae P	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Isostictidae	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	1.0
Leptoceridae L	0.0	0.0	5.0	0.0	4.5	3.8	0.0	4.5	80.0	100.0	175.0	150.0	62.5	18.8	12.5	2.4	0.0	13.6
Leptoceridae P	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0
Leptophlebiidae	169.0	110.5	57.5	13.0	18.2	15.0	0.0	0.0	0.0	0.0	0.0	0.0	637.5	87.5	56.3	40.5	108.0	145.5
Libellulidae	4.8	0.0	0.0	0.0	0.0	0.0	7.1	2.0	60.0	40.0	125.0	350.0	1.0	0.0	0.0	0.0	0.0	0.0
Mesoveliidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Naucoridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nematode	7.1	152.6	7.5	8.0	77.3	8.8	300.0	77.3	1140.0	700.0	1200.0	800.0	50.0	68.8	31.3	31.0	32.0	27.3
Nepidae	0.0	0.0	0.0	0.0	0.0	0.0	7.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Noteridae A	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Notonectidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oligochaeta	54.8	231.6	27.5	23.0	831.8	42.5	271.4	263.3	380.0	620.0	2225.0	3050.0	350.0	218.8	200.0	92.9	268.0	231.8
Oribatidae	11.9	21.1	2.5	2.0	0.0	0.0	200.0	131.8	1240.0	1560.0	1850.0	2250.0	87.5	37.5	12.5	11.9	4.0	22.7
Orthotrichia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ostracod	0.0	5.3	0.0	0.0	0.0	0.0	21.4	31.8	400.0	140.0	350.0	400.0	237.5	18.8	6.3	19.0	32.0	68.2
Palaemonidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Philopotomidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pleidae	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polycentropodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Protoneuridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pyralidae	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.5	0.0	6.3	0.0	0.0	0.0
Pyschodidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scirtidae	0.0	10.5	1.0	0.0	4.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4	4.0	0.0
Simuliidae L	0.0	5.3	2.5	0.0	9.1	3.8	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	1.0
Simuliidae P	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staphylinidae A	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staphylinidae L	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sundathelphusidae	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0
Tabanidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0
Tanypodinae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tipulidae	2.4	0.0	2.5	0.0	0.0	0.0	22.4	27.2	20.0	20.0	0.0	50.0	0.0	0.0	0.0	0.0	4.0	1.0
Unidentified	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	37.5	0.0	0.0	0.0	0.0	0.0
Veliidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Zygoptera Ind.	4.8	5.3	2.5	3.0	9.1	0.0	0.0	22.7	0.0	0.0	0.0	0.0	37.5	12.5	0.0	0.0	4.0	4.5

SD1			SD2			STW			7JD			7JU			CFD			CFU		
21/12/98			21/12/98			21/12/98			13/1/99			13/1/99			14/1/99			14/1/99		
1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
30.0	24.3	18.2	60.0	17.6	12.5	41.7	37.5	33.3	14.3	16.0	26.7	5.7	11.0	60.0	92.3	18.4	35.0	87.5	200.0	133.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	4.5	0.0	0.0	6.3	3.3	6.3	23.3	0.0	4.0	13.3	0.0	0.0	6.7	0.0	0.0	0.0	0.0	0.0	33.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	15.4	0.0	0.0	0.0	0.0	0.0
30.0	0.0	0.0	210.0	100.0	212.5	33.3	43.8	40.0	21.4	96.0	33.3	20.0	15.0	20.0	38.5	10.0	20.0	225.0	150.0	300.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	110.0	64.7	87.5	0.0	50.0	3.3	64.3	68.0	86.7	122.9	112.0	320.0	276.9	66.7	240.0	225.0	1600.0	1033.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	2.7	0.0	20.0	0.0	6.3	16.7	18.8	10.0	0.0	16.0	13.3	5.7	4.0	13.3	0.0	1.7	0.0	12.5	0.0	0.0
110.0	86.5	27.3	10.0	0.0	0.0	8.3	18.8	16.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0
60.0	43.2	22.7	160.0	35.3	56.3	191.7	75.0	86.7	164.3	80.0	126.7	42.9	7.0	120.0	176.9	53.3	50.0	112.5	550.0	533.3
2.0	2.7	9.1	10.0	11.8	37.5	0.0	0.0	0.0	28.6	8.0	13.3	5.7	4.0	26.7	7.7	0.0	10.0	0.0	50.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
270.0	94.6	18.2	50.0	29.4	162.5	266.7	31.3	30.0	278.6	40.0	106.7	8.6	0.0	33.3	23.1	3.3	0.0	200.0	950.0	500.0
0.0	2.7	0.0	0.0	11.8	1.0	0.0	0.0	0.0	7.1	0.0	0.0	1.0	0.0	1.0	15.4	1.7	0.0	12.5	50.0	0.0
10.0	13.5	22.7	0.0	5.9	0.0	8.3	18.8	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.0	0.0	0.0	0.0
30.0	0.0	4.5	20.0	341.2	50.0	150.0	37.5	6.7	7.1	0.0	0.0	0.0	0.0	0.0	7.7	0.0	0.0	125.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	5.7	2.0	6.7	0.0	0.0	0.0	0.0	0.0	33.3
0.0	2.7	13.6	0.0	0.0	0.0	8.3	25.0	13.3	7.1	0.0	0.0	0.0	10.0	6.7	330.8	76.7	205.0	200.0	50.0	300.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	10.8	18.2	0.0	0.0	6.3	0.0	0.0	3.3	21.4	0.0	0.0	2.9	1.0	0.0	0.0	1.7	10.0	0.0	0.0	33.3
10.0	2.7	4.5	20.0	11.8	12.5	8.3	12.5	6.7	7.1	4.0	6.7	0.0	3.0	6.7	46.2	13.3	55.0	125.0	200.0	133.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	0.0	5.7	1.0	0.0	7.7	1.7	15.0	12.5	0.0	33.3
110.0	2.7	13.6	300.0	17.6	12.5	0.0	0.0	0.0	7.1	24.0	13.3	17.1	10.0	80.0	76.9	10.0	30.0	62.5	100.0	66.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	2.7	0.0	0.0	17.6	6.3	0.0	0.0	0.0	0.0	0.0	6.7	5.7	0.0	60.0	0.0	1.7	1.0	12.5	50.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	8.3	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	5.4	13.6	0.0	0.0	0.0	0.0	18.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.5	0.0	0.0
20.0	0.0	4.5	10.0	0.0	0.0	16.7	6.3	16.7	0.0	0.0	0.0	1.0	0.0	6.7	7.7	0.0	15.0	50.0	100.0	200.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	14.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	6.7	0.0	0.0	5.0	0.0	100.0	33.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	11.8	1.0	0.0	0.0	0.0	7.1	1.0	0.0	0.0	3.0	0.0	1.0	0.0	5.0	0.0	0.0	0.0
0.0	0.0	4.5	70.0	5.9	12.5	8.3	6.3	0.0	0.0	8.0	0.0	28.6	78.0	133.3	46.1	6.7	50.0	37.5	150.0	166.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	50.0	0.0
50.0	40.5	90.9	120.0	358.8	381.3	0.0	0.0	0.0	14.3	64.0	126.7	37.1	3.0	13.3	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	15.4	0.0	1.0	2.0	2.0	1.0
0.0	0.0	1.0	0.0	0.0	0.0	16.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
80.0	0.0	5.5	0.0	0.0	0.0	30.0	93.8	40.0	100.0	12.0	53.3	100.0	3.0	206.7	415.4	143.3	340.0	500.0	1550.0	966.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	9.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1890.0	262.2	131.8	240.0	100.0	187.5	1950.0	668.8	376.7	257.1	164.0	246.7	91.4	4.0	180.0	353.7	63.5	105.0	1150.0	800.0	900.0
60.0	0.0	4.5	40.0	0.0	68.8	41.7	18.8	3.3	107.1	40.0	66.7	14.3	7.0	33.3	230.8	36.7	95.0	400.0	11450.0	3200.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	4.5	10.0	0.0	0.0	0.0	0.0	0.0	64.3	40.0	33.3	0.0	0.0	13.3	23.1	0.0	0.0	100.0	600.0	633.3
0.0	0.0	0.0	0.0	1.0	1.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	12.5	1.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.0	0.0	0.0	0.0
10.0	2.7	0.0	0.0	5.9	18.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0
0.0	2.7	4.5	0.0	0.0	0.0	8.3	0.0	3.3	7.1	0.0	0.0	0.0	0.0	6.7	1.0	0.0	1.0	0.0	0.0	33.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
10.0	10.8	13.6	10.0	0.0	25.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
90.0	62.2	50.0	770.0	35.3	93.8	0.0	0.0	0.0	7.1	8.0	13.3	8.6	1.0	0.0	0.0	0.0	0.0	12.5	0.0	0.0
0.0	0.0	0.0	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	2.7	4.5	0.0	0.0	0.0	8.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	2.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	8.3	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
20.0	2.7	9.1	20.0	17.6	0.0	25.0	37.5	30.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.0	33.3
10.0	1.0	4.5	1.0	0.0	0.0	0.0	0.0	0.0	0.0	12.0	0.0	0.0	0.0	0.0	0.0	0.0	5.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	2.7	13.6	0.0	0.0	6.3	0.0	0.0	0.0	14.3	4.0	0.0	2.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

NMD 13/1/99			NMU 13/1/99			SD1 12/1/99			SD2 12/1/99			SUM 12/1/99			STW 12/1/99			7JD 3/2/99		
1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
8.3	16.7	0.0	0.0	0.0	5.0	6.7	25.0	25.0	9.7	3.7	8.1	4.2	1.4	2.4	7.9	5.3	31.6	3.8	0.0	15.4
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	8.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	5.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	15.4
125.0	96.7	26.3	59.1	54.8	30.0	100.0	60.0	33.3	38.9	107.4	75.7	13.7	117.1	71.4	13.2	36.8	84.2	32.7	25.0	169.2
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
150.0	220.0	92.5	122.7	64.5	80.0	113.3	245.0	141.7	98.6	248.1	137.8	15.8	34.3	45.2	15.8	31.6	100.0	84.6	125.0	507.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.8	6.7	1.3	13.6	0.0	5.0	1.7	5.0	0.0	11.1	3.7	0.0	5.3	4.3	0.0	7.9	5.3	5.3	5.8	0.0	7.7
0.0	3.3	0.0	4.5	0.0	5.0	0.0	10.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	7.9	15.8	15.8	0.0	1.7	0.0
111.1	70.0	20.0	118.2	80.6	95.0	3.3	25.0	100.0	8.3	70.4	21.6	18.9	32.9	23.8	28.9	52.6	173.7	42.3	45.0	146.2
5.6	13.3	2.5	4.5	12.9	10.0	1.7	15.0	8.3	5.6	3.7	8.1	7.4	1.4	4.8	10.5	15.8	21.1	1.9	3.3	15.4
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	18.2	48.4	10.0	0.0	0.0	33.3	0.0	3.7	2.7	1.1	0.0	0.0	0.0	0.0	0.0	3.8	0.0	30.8
0.0	0.0	0.0	0.0	3.2	0.0	0.0	0.0	0.0	0.0	3.7	0.0	4.2	0.0	0.0	0.0	0.0	0.0	1.0	0.0	1.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	5.3	5.3	5.3	0.0	1.7	7.7
0.0	0.0	0.0	0.0	0.0	5.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	0.0	0.0	2.6	0.0	21.1	1.9	0.0	7.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.8	0.0	0.0	1.0	9.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	3.3	7.7
0.0	0.0	1.3	0.0	0.0	0.0	15.0	5.0	0.0	0.0	7.4	0.0	1.1	0.0	0.0	2.6	5.3	0.0	9.6	10.0	15.4
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.3	0.0	0.0	0.0
2.8	0.0	1.3	1.0	0.0	0.0	8.3	5.0	0.0	0.0	3.7	2.7	2.1	1.4	0.0	2.6	5.3	0.0	5.8	1.7	7.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	0.0	0.0	0.0	0.0	0.0	3.3	0.0
2.8	0.0	0.0	0.0	0.0	0.0	3.3	30.0	16.7	31.9	18.5	8.1	14.7	37.1	73.8	0.0	0.0	0.0	5.8	8.3	30.8
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	3.2	0.0	3.3	0.0	0.0	1.4	7.4	2.7	1.1	0.0	0.0	0.0	0.0	0.0	0.0	5.0	1.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	3.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.3	0.0	0.0	0.0	0.0	0.0
8.3	0.0	0.0	4.5	3.2	0.0	0.0	0.0	8.3	0.0	7.4	2.7	0.0	0.0	4.8	10.5	5.3	5.3	5.8	0.0	23.1
11.1	10.0	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8.3	6.7	0.0	0.0	6.5	5.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	7.9	5.3	21.1	3.8	1.7	7.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.0	0.0	0.0	0.0	10.0	1.0	0.0	0.0	0.0	7.4	5.0	4.2	4.3	0.0	0.0	0.0	0.0	0.0	0.0	1.0
25.0	6.7	18.8	68.2	77.4	35.0	26.7	25.0	83.3	15.3	14.8	18.9	3.2	10.0	16.7	42.1	42.1	52.6	1176.9	20.0	100.0
8.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.0	0.0	0.0	0.0	0.0
2.8	3.3	7.5	27.3	3.2	45.0	33.3	100.0	58.3	30.6	74.1	51.4	73.7	30.0	95.2	2.6	0.0	5.3	0.0	0.0	0.0
0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.0	5.3	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.3	0.0	0.0	0.0	0.0
19.4	23.3	3.8	68.2	32.3	80.0	11.7	110.0	125.0	1.7	85.2	3.2	0.0	0.0	21.4	50.0	115.8	94.7	17.3	21.7	84.6
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
100.0	56.7	33.8	268.2	158.1	225.0	16.7	55.0	91.7	56.9	222.2	83.8	29.5	31.4	140.5	384.2	800.0	236.8	50.0	26.7	246.2
5.6	6.7	3.8	18.2	12.9	20.0	0.0	20.0	58.3	12.5	7.4	86.5	1.1	0.0	0.0	21.1	78.9	21.1	7.7	13.3	84.6
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
13.9	0.0	0.0	4.5	6.5	35.0	0.0	5.0	8.3	0.0	7.4	0.0	0.0	0.0	0.0	42.1	36.8	0.0	1.9	0.0	7.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	1.0	0.0	5.3	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.7	8.1	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.8	0.0	0.0	0.0	0.0	0.0	0.0	5.0	0.0	0.0	3.7	0.0	0.0	0.0	0.0	2.6	0.0	0.0	0.0	1.7	7.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	1.0	5.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4	0.0	0.0	10.5	0.0	0.0	0.0
27.8	30.0	47.5	72.7	25.8	50.0	0.0	0.0	0.0	1.4	3.7	29.7	1.1	1.4	0.0	2.6	5.3	10.5	0.0	1.7	7.7
0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	8.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.0	0.0	0.0	0.0	0.0
0.0	1.0	0.0	0.0	0.0	5.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	1.3	4.5	0.0	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.3	1.4	7.4	2.7	2.1	4.3	31.0	0.0	5.3	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	3.3	0.0	0.0	6.5	5.0	0.0	10.0	0.0	5.6	11.1	2.7	0.0	0.0	0.0	0.0	0.0	0.0	3.8	3.3	15.4

7JU 3/2/99			CFD 4/2/99			CFU 4/2/99			NMD 3/2/99			NMU 3/2/99			SD1 2/2/99			SD2 2/2/99		
1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
18.6	6.0	8.1	55.0	4.5	36.4	50.0	280.0	175.0	2.3	2.0	3.2	15.0	48.0	54.0	16.0	18.0	2.4	8.0	2.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6	1.3	4.0	4.0	0.0	0.0	0.0	0.0	0.0	0.0
22.0	15.0	6.5	470.0	290.9	300.0	625.0	800.0	325.0	47.7	11.0	59.7	13.8	68.0	48.0	78.0	38.0	104.8	23.0	26.0	188.9
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
94.9	79.0	64.5	380.0	218.2	236.4	2800.0	1360.0	1525.0	61.4	32.0	85.5	78.8	184.0	94.0	76.0	132.0	142.9	34.0	8.0	222.2
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.7	0.0	9.7	5.0	4.5	0.0	0.0	0.0	0.0	0.0	2.0	3.2	1.3	1.0	6.0	4.0	0.0	2.4	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
13.6	8.0	25.8	15.0	13.6	27.3	75.0	240.0	200.0	43.2	28.0	35.5	41.3	44.0	52.0	30.0	18.0	16.7	40.0	2.0	37.0
3.4	0.0	0.0	10.0	1.0	27.3	0.0	0.0	25.0	2.3	9.0	1.6	5.0	8.0	10.0	2.0	8.0	2.4	4.0	1.0	7.4
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.7	1.0	4.8	0.0	13.6	4.5	0.0	0.0	25.0	0.0	0.0	1.6	0.0	4.0	14.0	0.0	0.0	0.0	3.0	0.0	0.0
0.0	0.0	0.0	0.0	2.0	1.0	25.0	0.0	0.0	2.3	0.0	0.0	2.5	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	2.0	0.0	0.0	0.0	3.7
0.0	0.0	0.0	0.0	4.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
10.2	19.0	1.6	100.0	40.9	36.4	250.0	60.0	50.0	0.0	0.0	0.0	2.5	0.0	2.0	8.0	16.0	0.0	16.0	0.0	3.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	18.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	1.6	5.0	4.5	9.1	25.0	60.0	5.0	2.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	0.0	0.0	10.0	13.6	9.1	100.0	60.0	50.0	0.0	1.0	1.6	1.3	1.0	2.0	2.0	4.0	2.4	0.0	0.0	3.7
0.0	0.0	0.0	0.0	4.5	0.0	0.0	0.0	0.0	0.0	0.0	1.6	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0
1.7	0.0	0.0	0.0	27.3	4.5	0.0	40.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	4.8	0.0	1.0	3.7
50.8	69.0	33.9	40.0	4.5	0.0	150.0	60.0	100.0	0.0	0.0	0.0	0.0	0.0	0.0	32.0	18.0	16.7	10.0	3.0	29.6
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	2.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	4.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3.4	5.0	1.6	5.0	0.0	0.0	25.0	20.0	25.0	2.3	0.0	1.6	2.5	0.0	2.0	0.0	4.0	0.0	1.0	0.0	3.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.7	1.0	0.0	5.0	4.5	4.5	50.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0	8.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	1.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0
6.8	19.0	9.7	5.0	0.0	4.5	150.0	60.0	25.0	25.0	12.0	25.8	40.0	96.0	50.0	68.0	34.0	59.5	11.0	1.0	37.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	3.7
1.7	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	25.0	6.0	4.8	0.0	0.0	0.0	18.0	4.0	28.6	6.0	0.0	25.9
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	3.0	0.0	0.0	0.0	8.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
0.0	0.0	0.0	0.0	0.0	0.0	25.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
27.1	8.0	17.7	35.0	190.9	309.1	30.0	98.0	1825.0	18.2	10.0	8.1	5.0	28.0	6.0	28.0	4.8	38.1	3.6	0.0	25.9
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
39.0	22.0	50.0	190.0	109.0	181.6	925.0	920.0	1700.0	68.2	31.0	30.6	51.3	72.0	36.0	66.0	16.0	31.0	48.0	32.0	140.7
11.9	8.0	9.7	5.0	104.5	50.0	350.0	740.0	1225.0	6.8	2.0	1.6	0.0	20.0	8.0	14.0	4.0	4.8	5.0	6.0	40.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	4.8	5.0	9.1	0.0	25.0	20.0	150.0	4.5	0.0	1.6	1.3	12.0	0.0	4.0	6.0	0.0	2.0	0.0	7.4
0.0	0.0	0.0	5.0	4.5	18.2	2.0	40.0	4.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.7	0.0	0.0	5.0	9.1	4.5	0.0	40.0	0.0	0.0	0.0	1.6	3.8	1.0	4.0	0.0	0.0	0.0	1.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	68.2	19.0	8.1	11.3	20.0	10.0	0.0	0.0	0.0	0.0	3.0	7.4
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.3	2.0	3.2	0.0	0.0	2.0	0.0	0.0	2.4	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	5.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.3	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	15.0	1.0	0.0	0.0	3.0	0.0	0.0	1.0	0.0	0.0	1.0	2.0	6.0	0.0	2.4	2.0	1.0	0.0
0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.7	2.0	0.0	0.0	4.5	4.5	0.0	0.0	0.0	6.8	0.0	1.6	0.0	0.0	4.0	0.0	2.0	2.4	0.0	0.0	3.7

SUM			STW			7JD			7JU			CFD			CFU			NMD		
2/2/99			2/2/99			23/2/99			23/2/99			24/2/99			24/2/99			23/2/99		
1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
2.0	6.0	14.0	1.0	0.0	0.0	53.3	16.7	61.9	16.2	2.9	3.8	6.1	0.0	10.0	62.5	260.0	225.0	27.3	32.4	2.9
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.0	0.0	0.0	0.0	0.0	2.4	0.0	0.0	0.0	20.0	0.0	0.0	0.0	0.0
1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	6.7	5.6	4.8	0.0	2.9	0.0	0.0	0.0	0.0	12.5	0.0	50.0	4.5	5.4	2.9
49.0	170.0	66.0	2.0	3.3	11.7	53.3	211.1	157.1	67.6	160.0	46.2	0.0	23.5	0.0	300.0	400.0	775.0	54.5	45.9	54.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
31.0	98.0	52.0	48.0	80.0	105.0	66.7	111.1	61.9	173.0	125.7	128.8	63.4	11.8	130.0	875.0	1600.0	2550.0	81.8	94.6	45.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	0.0	6.0	1.0	6.7	0.0	6.7	0.0	4.8	2.7	2.9	1.9	0.0	0.0	0.0	12.5	0.0	0.0	9.1	2.7	5.7
0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
13.0	18.0	34.0	64.0	45.0	86.7	253.3	188.9	133.3	40.5	34.3	11.5	9.8	11.8	80.0	100.0	240.0	350.0	204.5	127.0	85.7
2.0	6.0	2.0	5.0	1.7	6.7	6.7	16.7	4.8	2.7	14.3	1.9	2.4	0.0	0.0	12.5	40.0	50.0	18.2	5.4	5.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	4.0	0.0	0.0	0.0	6.7	11.1	14.3	5.4	0.0	0.0	2.4	0.0	0.0	12.5	0.0	0.0	36.4	0.0	2.9
0.0	2.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	2.0	6.0	0.0	2.0
0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	4.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	11.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	6.7	5.6	0.0	0.0	0.0	0.0	0.0	5.9	0.0	0.0	2.0	0.0	9.1	1.0	2.9
0.0	4.0	8.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.4	0.0	10.0	12.5	20.0	25.0	4.5	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.8	0.0	0.0	1.9	3.7	0.0	0.0	12.5	20.0	8.0	4.5	2.7	0.0
0.0	2.0	0.0	2.0	0.0	0.0	13.3	5.6	0.0	8.1	2.9	3.8	8.5	4.5	4.5	25.0	100.0	125.0	1.0	5.4	1.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	6.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0	0.0	50.0	80.0	125.0	0.0	0.0	0.0
36.0	26.0	16.0	4.0	10.0	5.0	0.0	5.6	9.5	8.1	0.0	11.5	3.7	0.0	20.0	87.5	300.0	425.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	0.0	2.0	0.0	0.0	0.0	1.0	0.0	4.8	21.6	0.0	9.6	0.0	0.0	0.0	1.0	3.0	0.0	9.1	2.7	5.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	2.0	2.0	0.0	0.0	0.0	0.0	5.6	9.5	0.0	0.0	1.0	0.0	0.0	0.0	12.5	0.0	50.0	4.5	8.1	5.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	50.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	9.0	6.7	10.0	13.3	5.6	28.6	0.0	2.9	0.0	0.0	0.0	0.0	0.0	20.0	25.0	9.1	8.1	0.0
0.0	0.0	0.0	1.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0
8.0	42.0	2.0	6.0	1.7	5.0	0.0	50.0	28.6	10.8	42.9	7.7	4.9	5.9	0.0	212.5	220.0	450.0	40.9	81.1	37.1
1.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6.0	10.0	4.0	1.0	0.0	0.0	0.0	0.0	0.0	5.4	0.0	3.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	1.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
14.0	4.8	7.2	7.0	6.0	0.0	626.7	83.3	214.3	81.1	82.9	51.9	47.6	111.8	12.0	350.0	780.0	1025.0	22.7	13.5	37.1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	17.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
44.0	28.0	152.0	40.0	135.0	50.0	140.0	155.6	85.7	67.6	37.1	34.6	58.5	211.8	120.0	150.0	60.0	75.0	181.8	86.5	57.1
0.0	2.0	8.0	0.0	5.0	8.3	73.3	50.0	42.9	21.6	8.6	13.5	8.5	76.5	160.0	275.0	1100.0	375.0	31.8	45.9	8.6
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	10.0	2.0	35.0	6.7	26.7	5.6	19.0	5.4	0.0	1.9	0.0	0.0	20.0	0.0	0.0	0.0	4.5	0.0	0.0
0.0	0.0	1.0	0.0	0.0	0.0	1.0	1.0	0.0	1.0	1.0	0.0	0.0	0.0	0.0	50.0	20.0	150.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	25.0	0.0	0.0	0.0
0.0	0.0	6.0	3.0	6.7	3.3	6.7	16.7	9.5	2.7	0.0	3.8	1.2	0.0	0.0	0.0	0.0	0.0	0.0	5.4	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	2.0	12.0	16.7	11.7	0.0	5.6	4.8	2.7	2.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.5	8.1	8.6
0.0	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	6.0	18.0	0.0	0.0	3.3	0.0	1.0	1.0	2.7	0.0	1.0	6.1	29.4	10.0	0.0	1.0	0.0	0.0	0.0	0.0
0.0	0.0	1.0	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	14.3	2.7	0.0	1.9	1.2	0.0	0.0	0.0	0.0	25.0	13.6	0.0	2.9

NMU			SD1			SD2			SUM			STW			7JD			7JU			CFD		
23/2/99			22/2/99			22/2/99			22/2/99			22/2/99			16/3/99			16/3/99			22/3/99		
1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
13.6	4.3	11.1	50.0	5.3	14.8	1.9	0.0	6.3	138.5	18.5	45.2	70.0	13.5	19.0	4.0	10.0	5.0	113.3	35.3	43.3	21.4	13.6	4.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	1.0	5.3	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	2.9	5.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	2.5	0.0	0.0	0.0	0.0	4.5	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
22.7	2.9	16.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
127.3	40.6	88.9	418.2	689.5	540.7	246.2	144.0	421.9	115.4	59.3	52.4	230.0	40.5	54.8	16.0	16.7	10.0	40.0	0.0	40.0	107.1	95.5	24.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
63.6	58.0	105.6	209.1	305.3	85.2	23.1	10.0	78.1	207.7	314.8	157.1	630.0	186.5	157.1	32.0	83.3	30.0	153.3	158.8	93.3	957.1	822.7	40.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.5	4.3	16.7	0.0	21.1	0.0	13.5	0.0	3.1	0.0	0.0	2.4	10.0	0.0	2.4	6.0	6.7	12.5	66.7	35.3	20.0	7.1	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
213.6	59.4	344.4	68.2	15.8	7.4	19.2	0.0	31.3	323.1	74.1	45.2	690.0	218.9	135.7	98.0	170.0	215.0	160.0	647.1	273.3	114.3	68.2	22.0
22.7	4.3	44.4	0.0	5.3	0.0	1.9	8.0	0.0	7.7	18.5	2.4	70.0	8.1	9.5	4.0	10.0	5.0	13.3	41.2	36.7	14.3	18.2	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
27.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	15.4	0.0	0.0	20.0	2.7	0.0	0.0	0.0	0.0	6.7	41.2	0.0	7.1	0.0	0.0
0.0	1.4	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	1.0	2.5	0.0	0.0	3.3	2.0	4.5	0.0
0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.5	0.0	5.6	0.0	0.0	3.7	0.0	0.0	0.0	0.0	0.0	0.0	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	92.9	22.7	6.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	2.9	5.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	9.1	0.0	3.7	0.0	0.0	0.0	0.0	7.4	2.4	20.0	0.0	0.0	2.0	3.3	0.0	6.7	17.6	33.3	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
9.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	17.6	0.0	0.0	0.0	0.0
0.0	2.9	5.6	1.0	21.1	3.7	3.8	2.0	0.0	7.7	1.0	7.1	1.0	5.4	2.4	0.0	0.0	0.0	0.0	5.9	0.0	21.4	18.2	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	9.1	10.5	11.1	3.8	2.0	3.1	0.0	22.2	9.5	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	14.3	22.7	0.0
0.0	0.0	0.0	72.7	110.5	37.0	15.4	2.0	12.5	30.8	166.7	59.5	0.0	0.0	4.8	0.0	3.3	0.0	26.7	0.0	0.0	64.3	72.7	6.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.3	0.0	6.7	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.5	0.0	5.6	0.0	5.3	0.0	0.0	0.0	0.0	7.7	7.4	2.4	0.0	0.0	0.0	2.0	0.0	7.5	13.3	0.0	6.7	0.0	9.1	2.0
4.5	11.6	5.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.5	4.3	11.1	0.0	5.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	110.0	24.3	71.4	6.0	0.0	0.0	6.7	0.0	0.0	7.1	9.1	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	4.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.4	1.0	0.0	0.0	0.0	0.0	0.0	0.0	7.7	2.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	13.3	0.0	0.0	0.0	0.0	0.0
40.9	15.9	11.1	118.2	31.6	29.6	19.2	3.0	9.4	0.0	63.0	66.7	60.0	10.8	4.8	8.0	3.3	12.5	106.7	41.2	23.3	64.3	63.6	4.0
0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.4	0.0	9.1	42.1	14.8	3.8	1.0	6.3	0.0	0.0	4.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.5	0.0	1.0	2.0	0.0	0.0	0.0	0.0	0.0	7.7	0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.0	0.0	0.0	1.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
59.1	2.9	50.0	16.4	0.0	0.0	6.9	7.0	0.0	18.5	26.7	5.7	108.0	6.5	8.6	0.0	0.0	9.0	8.0	21.2	0.0	42.9	86.4	4.8
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	7.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
195.5	13.0	188.9	27.3	31.6	7.4	32.7	2.0	43.8	38.5	3.7	35.7	260.0	5.4	38.1	72.0	256.7	75.0	186.7	100.0	0.0	14.3	22.7	24.0
68.2	0.0	27.8	4.5	0.0	0.0	15.4	0.0	15.6	38.5	14.8	16.7	70.0	0.0	2.4	4.0	13.3	2.5	86.7	23.5	30.0	107.1	63.6	34.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.5	0.0	5.6	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0	2.4	130.0	0.0	0.0	0.0	0.0	0.0	20.0	0.0	0.0	7.1	9.1	8.0
1.0	0.0	1.0	0.0	0.0	1.0	0.0	0.0	3.1	2.0	0.0	0.0	2.0	0.0	0.0	4.0	3.3	0.0	13.3	2.0	16.7	7.1	9.1	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.0	5.6	0.0	0.0	0.0	5.8	1.0	0.0	7.7	0.0	0.0	20.0	13.5	33.3	14.0	1.0	2.5	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.5	0.0	0.0	0.0	0.0	0.0	0.0
27.3	18.8	16.7	4.5	0.0	7.4	11.5	11.0	6.3	0.0	0.0	0.0	30.0	27.0	7.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	0.0	0.0	0.0	1.0	0.0	0.0	2.0	1.0	7.7	25.9	0.0	10.0	0.0	0.0	4.0	16.7	5.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	1.0	0.0	0.0	1.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.4	11.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.3	0.0	6.7	5.9	6.7	0.0	9.1	2.0

CFU			NMD			NMU			SD1			SD2			SUM			STW			7JD		
22/3/99			16/3/99			16/3/99			15/3/99			15/3/99			15/3/99			15/3/99			20/4/99		
1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
23.5	8.3	0.0	7.7	4.5	0.0	4.5	10.5	10.0	16.2	12.0	13.5	7.4	3.7	4.7	0.0	2.0	2.0	0.0	2.5	0.0	65.0	135.3	186.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	1.0	0.0	9.1	7.9	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	11.8	0.0
147.1	158.3	100.0	17.3	27.3	8.6	9.1	21.1	13.3	32.4	224.0	100.0	51.9	188.9	58.8	30.0	35.0	19.0	4.0	0.0	12.5	30.0	52.9	46.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
264.7	350.0	790.9	98.1	50.0	42.9	27.3	34.2	40.0	135.1	284.0	181.1	81.5	170.4	8.2	42.0	31.0	38.0	180.0	76.3	215.0	55.0	117.6	120.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
17.6	8.3	18.2	9.6	9.1	8.6	9.1	5.3	10.0	2.7	0.0	0.0	3.7	0.0	1.2	2.0	1.0	4.0	2.0	0.0	5.0	15.0	23.5	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	3.7	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
288.2	341.7	290.9	90.4	268.2	105.7	218.2	165.8	160.0	54.1	36.0	16.2	33.3	25.9	8.2	42.0	7.0	5.0	96.0	37.5	75.0	760.0	723.5	1073.3
11.8	16.7	18.2	9.6	22.7	28.6	0.0	15.8	10.0	8.1	0.0	18.9	7.4	0.0	1.2	4.0	1.0	3.0	8.0	2.5	5.0	20.0	23.5	20.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.5	0.0	0.0	0.0
0.0	41.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.9	0.0
5.9	1.0	0.0	1.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.9	25.0	9.1	1.9	4.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	1.9	0.0	0.0	1.0	2.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
5.9	8.3	9.1	0.0	0.0	0.0	4.5	0.0	0.0	8.1	4.0	8.1	3.7	0.0	0.0	4.0	1.0	0.0	0.0	1.3	0.0	25.0	64.7	13.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	25.0	5.9	0.0
11.8	25.0	9.1	0.0	0.0	2.9	0.0	2.6	0.0	0.0	0.0	2.7	0.0	0.0	0.0	1.0	1.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.0	15.0	11.8	1.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	2.0	0.0	0.0	0.0	0.0	2.5	0.0	0.0	0.0
5.9	0.0	36.4	0.0	0.0	0.0	0.0	2.6	0.0	10.8	12.0	8.1	11.1	122.2	5.9	6.0	5.0	6.0	2.0	1.3	0.0	5.0	0.0	13.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.9	0.0	0.0	1.9	4.5	0.0	0.0	1.0	3.3	0.0	0.0	0.0	0.0	3.7	3.5	0.0	0.0	0.0	0.0	0.0	0.0	5.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	9.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	1.0	1.0	4.5	15.8	0.0	2.7	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	10.0	0.0	0.0
0.0	0.0	0.0	0.0	9.1	1.0	4.5	0.0	0.0	0.0	4.0	1.0	3.7	3.7	0.0	0.0	0.0	0.0	4.0	1.3	1.0	0.0	5.9	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
23.5	0.0	18.2	5.8	31.8	2.9	4.5	5.3	26.7	5.4	4.0	5.4	3.7	3.7	0.0	0.0	0.0	0.0	10.0	12.5	22.5	0.0	88.2	153.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.9	8.3	18.2	19.2	36.4	28.6	22.7	21.1	16.7	110.8	152.0	48.6	59.3	151.0	27.1	26.0	16.0	13.0	2.0	11.3	10.0	30.0	117.6	260.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	2.7	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
70.6	1700.0	945.5	40.4	72.7	22.9	286.4	23.7	60.0	22.7	32.0	0.0	111.1	107.4	0.0	100.0	29.0	21.0	9.6	9.0	33.0	100.0	105.9	20.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.0	0.0	0.0
441.2	316.7	472.7	32.7	154.5	40.0	181.8	126.3	113.3	37.8	96.0	81.1	70.4	66.7	15.3	16.0	4.0	1.0	58.0	6.3	47.5	30.0	47.1	13.3
164.7	200.0	381.8	3.8	13.6	11.4	9.1	7.9	6.7	13.5	28.0	5.4	40.7	85.2	1.2	20.0	0.0	0.0	8.0	1.3	15.0	75.0	41.2	53.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	9.1	1.9	4.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.0	2.5	22.5	5.0	0.0	0.0
23.5	5.0	9.1	0.0	2.0	0.0	1.0	0.0	6.7	0.0	0.0	2.0	1.0	1.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	1.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	1.0	0.0	0.0	2.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	1.0	3.0	16.0	15.0	17.5	0.0	5.9	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	1.9	13.6	0.0	0.0	0.0	0.0	0.0	12.0	0.0	0.0	3.7	2.4	0.0	0.0	0.0	5.0	6.3	15.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	7.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	1.0	0.0	4.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	6.3	1.0	3.8	0.0	0.0	0.0	0.0	0.0	21.6	40.0	35.1	22.2	0.0	1.2	2.0	3.0	2.0	2.0	1.3	0.0	0.0	0.0	1.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.9	25.0	18.2	0.0	0.0	0.0	4.5	7.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	5.0	5.0	0.0

7JU			CFD			CFU			NMD			NMU			SD1			SD2		
20/4/99			4/23/99			4/23/99			20/4/99			20/4/99			19/4/99			19/4/99		
1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
70.6	93.8	136.4	41.7	1100.0	400.0	166.7	200.0	233.3	11.8	22.7	28.0	15.6	23.4	4.8	77.3	50.0	72.7	212.5	211.1	20.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	17.6	4.5	52.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	18.2	0.0	200.0	700.0	0.0	0.0	1.0	0.0	0.0	4.0	0.0	0.0	4.8	0.0	0.0	4.5	0.0	0.0	0.0
0.0	0.0	9.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	9.1	1.0	0.0	0.0	0.0	0.0	0.0	29.4	9.1	12.0	1.0	6.4	1.0	0.0	0.0	0.0	0.0	0.0	0.0
135.3	112.5	454.5	33.3	2100.0	3500.0	266.7	500.0	566.7	17.6	27.3	68.0	25.0	8.5	26.2	81.8	210.0	163.6	500.0	333.3	90.0
0.0	0.0	0.0	1.0	0.0	0.0	5.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
117.6	181.3	63.6	200.0	8600.0	47600.0	900.0	2600.0	1166.7	58.8	45.5	60.0	21.9	23.4	35.7	250.0	200.0	263.6	675.0	844.4	270.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.9	12.5	36.4	25.0	200.0	100.0	3.0	0.0	0.0	41.2	9.1	12.0	18.8	14.9	0.0	0.0	5.0	4.5	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	100.0	0.0	0.0	0.0	0.0	0.0	4.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
288.2	531.3	372.7	608.3	1700.0	12200.0	4266.7	7900.0	2333.3	552.9	427.3	624.0	265.6	221.3	140.5	95.5	85.0	54.5	525.0	766.7	600.0
17.6	31.3	9.1	66.7	300.0	1300.0	100.0	400.0	166.7	17.6	31.8	16.0	3.1	8.5	7.1	27.3	20.0	9.1	137.5	100.0	50.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
11.8	31.3	0.0	8.3	1100.0	100.0	66.7	0.0	133.3	11.8	13.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	1.0	1.0	100.0	0.0	2.0	1.0	29.4	22.7	20.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.1	0.0	0.0	18.2	5.0	0.0	0.0	0.0	0.0
5.9	6.3	0.0	0.0	0.0	0.0	0.0	0.0	33.3	0.0	4.5	0.0	0.0	2.1	0.0	0.0	0.0	0.0	0.0	11.1	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	1.0	3.1	1.0	2.4	0.0	0.0	1.0	0.0	0.0	0.0
29.4	81.3	118.2	25.0	3100.0	900.0	0.0	100.0	33.3	11.8	13.6	4.0	0.0	2.1	0.0	4.5	0.0	1.0	0.0	0.0	90.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.9	0.0	0.0	8.3	0.0	0.0	0.0	0.0	0.0	23.5	1.0	0.0	0.0	1.0	0.0	22.7	0.0	4.5	0.0	11.1	0.0
0.0	0.0	0.0	8.3	900.0	2000.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	9.1	5.0	40.9	0.0	0.0	10.0
0.0	0.0	9.1	0.0	100.0	1300.0	33.3	0.0	0.0	11.8	0.0	4.0	3.1	2.1	0.0	0.0	0.0	4.5	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	200.0	166.7	100.0	66.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	30.0	0.0	0.0	0.0	0.0
23.5	25.0	18.2	25.0	700.0	4400.0	433.3	2500.0	200.0	0.0	0.0	0.0	0.0	0.0	0.0	9.1	10.0	9.1	25.0	22.2	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0	0.0
1.0	6.3	0.0	3.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1	1.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	8.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	6.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	0.0	0.0	0.0	0.0	0.0	33.3	4.0	0.0	5.9	1.0	0.0	0.0	2.1	2.4	1.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	7.1	0.0	10.0	0.0	37.5	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
11.8	18.8	18.2	8.3	0.0	2200.0	0.0	0.0	0.0	11.8	0.0	32.0	46.9	4.3	33.3	9.1	30.0	22.7	100.0	133.3	20.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	9.4	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
88.2	37.5	163.6	108.3	500.0	1500.0	800.0	800.0	533.3	35.3	22.7	52.0	31.3	10.6	21.4	213.6	285.0	290.9	287.5	211.1	130.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.4	4.5	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
5.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.5	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
14.1	0.0	10.9	350.0	240.0	800.0	360.0	5600.0	1133.3	52.9	36.4	16.0	15.6	12.8	4.8	10.9	0.0	5.5	30.0	77.8	36.0
0.0	0.0	1.0	2.0	0.0	0.0	0.0	0.0	0.0	11.8	1.0	4.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
41.2	6.3	45.5	83.3	100.0	200.0	33.3	100.0	366.7	76.5	50.0	12.0	21.9	17.0	16.7	13.6	5.0	9.1	37.5	77.8	160.0
170.6	137.5	263.6	116.7	3800.0	5600.0	566.7	3100.0	833.3	29.4	22.7	40.0	34.4	17.0	11.9	13.6	5.0	31.8	212.5	55.6	20.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	9.1	0.0	300.0	300.0	0.0	0.0	0.0	17.6	4.5	0.0	3.1	0.0	0.0	0.0	0.0	0.0	12.5	0.0	0.0
1.0	0.0	27.3	7.0	3.0	100.0	33.3	100.0	33.3	5.9	4.5	8.0	6.3	0.0	2.4	4.0	1.0	4.5	37.5	33.3	50.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	8.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	1.0	0.0	0.0	0.0
5.9	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	11.8	0.0	0.0	12.5	6.4	33.3	4.5	5.0	4.5	37.5	0.0	2.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.3	0.0	7.1	4.5	45.0	9.1	12.5	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	8.3	200.0	0.0	1.0	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.6	10.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0	0.0	0.0	0.0
35.3	50.0	18.2	8.3	1700.0	3000.0	300.0	700.0	133.3	52.9	31.8	16.0	0.0	0.0	0.0	4.5	0.0	0.0	0.0	0.0	10.0

SUM 19/4/99			STW 19/4/99			7JD 11/5/99			7JU 11/5/99			CFD 12/5/99			CFU 12/5/99			NMD 11/5/99		
1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
46.2	65.4	51.4	133.3	37.5	125.0	88.9	66.7	12.5	137.5	154.5	62.5	125.0	0.0	60.0	10.0	0.0	30.0	0.0	5.9	10.6
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0	0.0	0.0	0.0	0.0	25.0	375.0	60.0	0.0	28.6	20.0	0.0	2.9	0.0
5.8	1.9	0.0	0.0	12.5	0.0	22.2	16.7	0.0	2.0	9.1	37.5	0.0	0.0	0.0	0.0	0.0	0.0	3.7	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
13.5	26.9	10.8	133.3	75.0	100.0	22.2	0.0	125.0	125.0	81.8	75.0	150.0	100.0	160.0	20.0	28.6	140.0	48.1	17.6	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0
32.7	46.2	45.9	900.0	500.0	187.5	222.2	58.3	237.5	250.0	290.9	350.0	1100.0	1050.0	980.0	10.0	42.9	80.0	29.6	20.6	25.5
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3.8	0.0	2.7	16.7	25.0	50.0	77.8	25.0	137.5	87.5	18.2	75.0	150.0	200.0	120.0	0.0	57.1	80.0	25.9	50.0	8.5
0.0	0.0	0.0	0.0	0.0	0.0	33.3	8.3	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	10.0	0.0	2.9	0.0
59.6	88.5	91.9	1150.0	887.5	1362.5	1122.2	625.0	1587.5	575.0	518.2	1012.5	2875.0	1575.0	1420.0	1060.0	1085.7	1400.0	351.9	261.8	97.9
5.8	7.7	21.6	33.3	37.5	150.0	111.1	75.0	150.0	112.5	90.9	25.0	25.0	100.0	140.0	20.0	42.9	120.0	29.6	2.9	2.1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	25.0	0.0	8.3	0.0	0.0	0.0	0.0	100.0	25.0	0.0	0.0	14.3	20.0	0.0	0.0	0.0
1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.5	1.0	25.0	20.0	0.0	0.0	10.0	3.7	0.0	2.1
0.0	11.5	0.0	0.0	0.0	0.0	11.1	16.7	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	25.0	0.0	0.0	0.0	0.0	0.0	0.0	25.0	0.0	0.0	20.0	14.3	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	25.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	25.0	0.0	80.0	10.0	0.0	10.0	0.0	0.0	2.1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	11.1	16.7	0.0	12.5	9.1	0.0	0.0	0.0	0.0	0.0	0.0	60.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1
13.5	0.0	5.4	0.0	0.0	0.0	11.1	16.7	12.5	112.5	109.1	25.0	0.0	125.0	40.0	20.0	1.0	0.0	40.7	38.2	46.8
3.8	1.9	2.7	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.9	0.0	16.7	25.0	50.0	77.8	25.0	75.0	150.0	181.8	150.0	50.0	1.0	40.0	0.0	0.0	0.0	7.4	5.9	0.0
0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0	12.5	0.0	9.1	0.0	0.0	0.0	0.0	10.0	0.0	10.0	0.0	0.0	0.0
13.5	11.5	0.0	0.0	0.0	0.0	0.0	8.3	0.0	12.5	9.1	12.5	0.0	25.0	20.0	0.0	0.0	10.0	7.4	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.9	5.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.7	0.0	2.1
1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	2.0	1.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
3.8	1.9	13.5	0.0	0.0	0.0	22.2	41.7	37.5	100.0	154.5	162.5	0.0	0.0	0.0	0.0	0.0	0.0	18.5	8.8	10.6
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	8.1	0.0	0.0	0.0	100.0	108.3	112.5	50.0	45.5	25.0	0.0	0.0	0.0	0.0	0.0	0.0	14.8	23.5	14.9
15.4	7.7	5.4	0.0	0.0	0.0	0.0	8.3	0.0	0.0	27.3	0.0	0.0	1.0	1.0	0.0	0.0	10.0	11.1	8.8	0.0
0.0	1.9	0.0	50.0	125.0	12.5	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	12.5	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.9	0.0	0.0	616.7	312.5	187.5	44.4	16.7	237.5	137.5	118.2	75.0	75.0	100.0	60.0	0.0	0.0	0.0	14.8	2.9	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.7	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
101.9	113.5	48.6	83.3	75.0	12.5	77.8	25.0	137.5	125.0	218.2	237.5	250.0	400.0	660.0	10.0	0.0	60.0	22.2	0.0	6.4
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1
0.0	0.0	0.0	0.0	0.0	9.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0	12.5	0.0	0.0	12.5	2.0	1.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	122.2	83.3	112.5	50.0	45.5	12.5	0.0	0.0	0.0	0.0	14.3	0.0	7.4	11.8	12.8
0.0	0.0	0.0	0.0	0.0	0.0	11.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
4.6	6.9	18.9	166.7	287.5	587.5	40.0	20.0	0.0	15.0	81.8	15.0	425.0	750.0	120.0	1020.0	1428.6	260.0	17.8	0.0	0.0
0.0	0.0	0.0	1.0	12.5	0.0	0.0	1.0	1.0	3.0	9.1	0.0	8.0	5.0	7.0	10.0	0.0	2.0	3.7	2.9	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0
3.8	3.8	2.7	66.7	100.0	50.0	22.2	75.0	0.0	37.5	72.7	12.5	0.0	25.0	0.0	130.0	185.7	50.0	33.3	26.5	25.5
5.8	28.8	5.4	116.7	175.0	12.5	122.2	0.0	50.0	25.0	0.0	0.0	200.0	175.0	20.0	120.0	171.4	0.0	22.2	14.7	2.1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.9	0.0	33.3	12.5	12.5	0.0	0.0	0.0	0.0	9.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	3.8	5.4	1.0	2.0	13.0	22.2	16.7	37.5	3.0	9.1	11.0	2.0	3.0	2.0	20.0	13.0	60.0	5.0	2.9	2.1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.7	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	7.4	5.9	2.1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	25.0	3.0	0.0	0.0	0.0	0.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.9	2.7	133.3	212.5	12.5	0.0	0.0	12.5	1.0	0.0	0.0	25.0	0.0	0.0	0.0	28.6	0.0	11.1	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	16.7	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	3.7	0.0	0.0
0.0	3.8	0.0	216.7	75.0	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	16.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	8.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	1.0	0.0	0.0	0.0	1.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	3.8	2.7	0.0	0.0	1.0	77.8	191.7	12.5	2.0	9.1	62.5	0.0	0.0	0.0	0.0	0.0	10.0	7.4	17.6	2.1
0.0	0.0	0.0	0.0	0.0	0.0	11.1	8.3	12.5	12.5	9.1	0.0	0.0	0.0	0.0	30.0	0.0	0.0	0.0	5.9	2.1
0.0	0.0	0.0	0.0	0.0	0.0	33.3	66.7	37.5	12.5	18.2	12.5	0.0	0.0	0.0	0.0	14.3	0.0	3.7	0.0	0.0
1.9	0.0	0.0	0.0	0.0	0.0	0.0	16.7	12.5	162.5	109.1	150.0	25.0	125.0	60.0	10.0	14.3	150.0	33.3	50.0	21.3

NMU 11/5/99			SD1 10/5/99			SD2 10/5/99			SUM 10/5/99			STW 10/5/99		
1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
2.8	9.4	18.5	116.7	76.5	125.0	175.0	78.4	53.8	26.9	17.3	32.7	47.1	111.1	240.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.4	0.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	1.9	3.8	0.0	23.5	0.0	0.0
1.4	3.1	0.0	0.0	0.0	16.7	12.5	0.0	3.8	3.8	1.9	0.0	0.0	0.0	0.0
2.8	9.4	25.9	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
31.9	31.3	74.1	50.0	23.5	33.3	162.5	48.6	7.7	0.0	0.0	0.0	52.9	55.6	80.0
0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8.3	28.1	22.2	666.7	147.1	233.3	475.0	118.9	50.0	9.6	11.5	1.9	23.5	133.3	320.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0
6.9	37.5	25.9	166.7	29.4	33.3	25.0	10.8	13.5	3.8	0.0	9.6	29.4	100.0	40.0
0.0	3.1	0.0	16.7	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0
95.8	243.8	366.7	1233.3	464.7	808.3	1262.5	486.5	223.1	238.5	223.1	251.9	700.0	1466.7	2800.0
4.2	12.5	11.1	216.7	41.2	50.0	150.0	37.8	21.2	15.4	3.8	25.0	23.5	88.9	80.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	1.9	29.4	0.0	0.0
0.0	0.0	3.7	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.4	0.0	0.0	16.7	0.0	0.0	0.0	0.0	1.9	5.8	3.8	0.0	0.0	0.0	0.0
0.0	0.0	0.0	16.7	0.0	8.3	0.0	0.0	0.0	0.0	0.0	0.0	58.8	0.0	420.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0
1.4	3.1	0.0	0.0	0.0	0.0	12.5	2.7	1.9	0.0	0.0	0.0	0.0	11.1	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.9	5.8	5.8	11.8	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0
0.0	0.0	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.8	12.5	3.7	166.7	123.5	58.3	25.0	10.8	5.8	1.9	17.3	3.8	88.2	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.0
13.9	12.5	11.1	16.7	29.4	50.0	75.0	32.4	40.4	0.0	1.9	1.9	4.0	0.0	1.0
0.0	0.0	0.0	0.0	0.0	8.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	16.7	0.0	16.7	62.5	8.1	9.6	0.0	1.9	3.8	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	7.4	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	5.8	0.0	0.0	0.0
0.0	0.0	0.0	0.0	1.0	0.0	1.0	2.7	0.0	0.0	0.0	0.0	1.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.4	3.1	3.7	50.0	5.9	16.7	0.0	2.7	7.7	0.6	117.3	7.7	41.2	0.0	0.0
0.0	0.0	0.0	16.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	1.0	0.0	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2.8	6.3	11.1	166.7	5.9	8.3	37.5	10.8	9.6	13.5	19.2	5.8	1.0	11.1	0.0
1.4	9.4	0.0	0.0	0.0	8.3	0.0	0.0	0.0	7.7	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	16.7	5.9	16.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	22.2	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
13.9	12.5	22.2	100.0	47.1	66.7	62.5	13.5	13.5	1.9	0.0	0.0	23.5	222.2	80.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
12.5	15.6	33.3	16.7	52.0	33.3	100.0	51.4	59.6	17.3	40.4	38.5	0.0	22.2	20.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	3.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	2.0	0.0	0.0	25.0	0.0	0.0	0.0	0.0	0.0	11.8	0.0	0.0
0.0	15.6	3.7	16.7	5.9	0.0	0.0	0.0	0.0	3.8	0.0	3.8	0.0	11.1	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.7	11.3	8.9	20.0	0.0	50.0	0.0	0.0	0.0	2.3	26.9	0.0	129.4	100.0	560.0
0.0	1.0	1.0	10.0	0.0	8.0	12.5	1.0	1.9	2.0	0.0	0.0	0.0	1.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6.9	3.1	0.0	66.7	47.1	16.7	25.0	0.0	17.3	0.0	5.8	5.8	76.5	44.4	120.0
1.4	3.1	22.2	50.0	11.8	33.3	100.0	2.7	5.8	0.0	17.3	9.6	64.7	11.1	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	11.8	0.0	20.0
6.9	9.4	11.1	4.0	1.0	16.7	25.0	5.4	1.9	0.0	2.0	0.0	5.9	9.0	3.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.0	0.0	0.0	2.0	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	7.7	1.9	0.0	0.0	0.0	22.2	0.0
0.0	0.0	0.0	0.0	0.0	8.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	3.1	0.0	16.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	11.1	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0
0.0	3.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.0	9.4	0.0	183.3	0.0	8.3	0.0	2.7	0.0	1.9	3.8	1.9	0.0	11.1	0.0
1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0
0.0	12.5	3.7	83.3	5.9	8.3	25.0	0.0	3.8	5.8	1.9	9.6	11.8	0.0	0.0
9.7	9.4	22.2	116.7	58.8	133.3	112.5	70.0	36.5	15.4	19.2	19.2	5.9	0.0	20.0