Physicochemical variable in June, 1996

Water t	Velo	Width	Depth	Discha	pH	Alka	Condu	TDS	Turbid	TSS	Or-P	N	DO	BOD
33.10	0.15	32.00	4.02	18.88	8.40	62.00	288.83	192.17	430.00	218.00	0.04	0.30	5.80	0.52
31.43	0.16	31.00	3.97	19.67	7.97	64.00	321.50	214.50	400.00	153.00	0.04	0.40	5.95	1.02
31.33	0.32	23.00	1.79	12.03	8.05	82.00	353.17	235.33	395.00	165.50	0.04	0.40	6.50	0.89
23.50	0.43	5.60	0.14	0.34	8.25	72.00	141.78	94.38	21.00	9.00	0.00	0.10	6.91	0.69
24.00	0.28	4.30	0.21	0.25	8.20	36.00	62.43	41.60	62.00	31.50	0.00	0.10	6.51	1.25
24.30	0.37	6.65	0.35	0.86	7.25	90.00	141.63	93.83	41.00	17.00	0.04	0.20	6.80	0.96
28.15	0.33	3.20	0.10	0.10	7.73	92.00	187.78	125.15	7.50	3.50	0.02	0.30	7.33	0.57
28.68	1.34	14.60	0.61	11.98	8.13	106.00	224.33	150.02	125.00	253.00	0.00	0.10	6.88	1.30
28.13	0.40	19.00	2.19	16.61	8.07	92.00	181.18	120.75	80.00	93.00	0.00	0.20	7.40	0.75
28.35	0.37	27.00	2.07	20.85	8.02	92.00	187.42	124.90	72.00	89.00	0.00	0.20	7.30	1.35
28.93	0.29	24.00	3.20	22.33	8.25	108.00	229.17	152.60	53.00	63.00	0.00	0.20	7.03	2.50
29.08	0.24	24.00	2.75	15.82	8.20	96.00	194.45	129.60	81.00	73.00	0.04	0.30	6.52	1.12
29.68	1.04	12.00	0.38	4.68	7.43	56.00	120.20	80.20	200.00	228.50	0.02	0.30	7.20	0.96
30.00	0.52	5.00	0.22	0.56	7.72	202.00	333.00	221.30	26.00	23.50	0.02	0.70	5.63	0.94
28.25	0.75	12.00	0.76	6.86	7.87	86.00	158.88	105.80	100.00	212.50	0.00	0.30	6.63	3.31
29.20	0.44	22.00	0.35	3.42	8.18	98.00	174.40	115.92	350.00	305.00	0.14	0.40	6.78	1.92
29.58	0.13	32.00	2.29	9.77	8.10	74.00	146.15	97.27	128.00	274.50	0.02	1.70	5.08	3.26
29.30	0.31	3.70	0.82	1.43	8.50	102.00	232.40	137.30	15.00	13.00	0.02	0.30	6.30	1.22
28.40	0.18	29.00	0.69	3.60	8.60	126.00	287.40	193.90	23.50	32.00	0.30	0.70	3.20	10.50
29.10	0.12	58.00	1.20	8.35	8.60	130.00	311.20	222.13	24.00	38.50	0.40	1.10	2.80	12.80
29.58	0.39	39.00	3.01	69.26	8.08	94.00	172.03	115.02	27.00	23.50	0.04	0.70	4.72	1.55
31.18	0.43	40.00	4.74	82.36	8.10	78.00	191.56	126.44	33.00	22.50	0.02	0.80	6.88	1.94
28.00	0.24	53.00	4.20	46.48	8.20	90.00	178.90	122.60	27.50	30.00	0.02	0.20	6.10	1.41
30.00	0.28	68.00	7.10	117.61	8.20	92.00	172.70	116.80	8.50	8.50	0.00	0.10	3.80	1.32
29.00	0.27	69.00	7.70	124.80	8.30	98.00	174.10	118.20	7.80	6.30	0.00	0.20	5.30	1.68
28.00	0.19	49.00	2.40	19.44	8.20	102.00	169.40	102.70	6.50	8.00	0.02	0.50	5.80	1.70
28.20	0.16	42.00	3.10	29.82	8.50	86.00	172.00	115.20	13.00	12.50	0.00	0.60	5.70	1.84

The performance of biotic scores and indices in assessing water pollution: a case study in the Pong catchment of north-east Thailand.

Abstract

Estimates of family diversity were made at the scale of the catchment, three regions along the catchment (upper, mid and lower) and at individual sites. Both methods yielded similar estimates of whole catchment family diversity (31-34 families). However this differed at the scale of the regions within catchments for streambed grab samples (4.1-12.3 families) but not edgewater samples (7.3-8.1 families), and at the scale of the sites within the regions.

Families were unevenly sampled by the 2 methods at each site. Gerridae, Baetidae, Corixidae and Protoneuridae dominated the edge samples, while streambed grab samples were rich in Chironomidae, Hydropsychidae and Corbiculidae.

These results suggest that scale is an important consideration when determining an appropriate sampling protocol.

Among indices and scores tested measures of species richness, family richness, and EPT richness best-reflected water quality. Shannon-Wiener index most significantly correlated to water pollution. BMWP/ASPT system was significantly correlated to organic water pollution.

Introduction

The assessment of water quality using benthic macroinvertebrate data is currently applied via two main approaches; firstly, by applying various indices and score systems, and secondly, by employing multivariate analyses of community structure. The index and score systems (sometimes called "metrics", particularly in North America) are more popularly used among water authorities in continental Europe and North America (Johnson *et al.* 1993).

The index systems were first developed and derived from the classical German method-the Saprobien system invented by Kolkwitz and Marsson in 1909 (Metcalfe 1989). Water quality according to this system is classified into polysaprobic, alpha- and beta-mesosaprobic and oligosaprobic zones in which each water body, respectively, ranges from highly polluted to saturated oxygen with very diverse fauna. The indicator taxa used in the Saprobien system are mainly bacteria, algae, protozoan, rotifers and some benthic macroinvertebrates.

Later, the Trent Biotic Index (TBI) was created by Woodiwiss in 1964 for monitoring water quality in the United Kingdom (Johnson *et al.* 1993). This index is mainly derived from the Saprobien system, but it focuses on using benthic macroinvertebrates as indicator taxa. Modified versions of these two systems are now in use at regional and local levels throughout Europe.

However, the above index systems have a number of limitations. Firstly, they are effective only in local geographical areas, and secondly, taxa identification usually requires expert personnel. As an alternative to the index system, the score system was later developed. The first era of the score system is marked by the development of Chandler's Biotic Score (CBS) system in Scotland (Chandler 1970), followed by

Chutter (1972) who proposed the use of a scoring system (The Chutter Score) for assessing water quality in South Africa. These two scoring methods were the first systems which exclusively used benthic macroinvertebrates for assessing water quality.

Still, with the above two score methods, the indicator taxa inevitably require species level identification, and this only apply to the species which existed in a certain locality. Their later use are still in question, for example, the CBS tested and found by Able (1989) that it was effective in detecting organic water pollution, while Pinder and Farr (1987) determined it to be insensitive. Research biologists focused more on the necessity of high taxonomic resolution for more precise inferential information in assessing water pollution. Water quality managers, in contrast, require rather rapid biological methods (like the chemical) which are time-efficient and cost-effective.

To meet the above gap, the Biological Monitoring Working Party system (BMWP) was then invented. This score system uses benthic macroinvertebrate taxa identified at family level (Armitage *et al.* 1983). Each macroinvertebrate family is assigned the score according to its relative tolerance value (the score ranges 0-10). Later, the BMWP score was discovered to vary less with season and sampling methods if divided by the taxa (family) numbers, and become the Average Score per Taxon (ASPT). The BMWP was found to be more sensitive in detecting organic pollution, but less sensitive in unpolluted waters (Bargos *et al.* 1990).

The BMWP is still in use in the United Kingdom and has influenced the shortcoming of the score systems. The Hilsenhoff's Family-Level Biotic Index is widely used in North America (Hilsenhoff 1988). In Australia, the SIGNAL (Stream Invertebrate Grade Number-Average Level) was proposed by Chessman (1995). The first method uses only arthropod taxa while the second takes into account all macroinvertebrate groups. Both methods are designed for rapid bioassessment of water quality. The latter was recently tested, and was found to be quite promising (Growns *et al.* 1995). However, these two scoring methods require more tests conducted in other "rivers" apart from "shallow streams" in mountainous areas, for example, the deeper rivers in tropical climates. Consequently, these score systems will be tested in this study.

Unlike the score system which was primarily based on pollution tolerant values, the indices system (sometimes called community structure indices) emphasises on the variation of a species in a community sample. Washington (1984) reviewed eighteen indices most commonly used (e.g., Shannon's, Magalef's, Simpson's, Menhinick's etc.) and he found that each index has limited uses, and all indices require extensively tests with respect to, for example, seasonal regime, sampling methods, sample size, duration of sampling and taxonomic level. Among those indices, even the most recommended index-the Shannon-Wiener Index (UNEP, WHO, UNESCO and WMO, 1992) for Washington (1984) to be less relevant noted use in water quality assessment. Surprisingly, indeed, current publications about the performance of these indices in evaluating water quality are rare.

In addition to the above score and indices systems, the multimetric (multiple indices) was now recently proposed for use in the United States (see more details in Resh *et al.* 1995 and Barbour *et al.* 1995 and references therein). This new approach was influenced by the rapid bioassessment method created by Plafkin *et al.* (1989). The multimetric approach was developed as to rectify the former individual metrics which many claimed to have limitations in use (e.g., as in Norris and George 1993). In fact, the multimetric approach to date is the combination of the former indices traditionally used. Its aim is to reduce the weakness or increase the strength of individual indices by combining them together. It is also claimed that multiple indices can contribute more meaningful and effective ecological information for water resource planning (Barbour *et al.* 1995).

To date, the most commonly used multimetrics are richness measures, including total species richness and Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa abundance (see details in Barbour *et al.* 1995). In addition to the EPT, individual metrics for Ephemeroptera, Trichoptera or Diptera abundance was also been adopted (DeShon 1995). All these are based on the conventional notion that degradation of water quality will also reduce the number of benthic species, particularly sensitive taxa (Spellerberg 1992, Resh 1995 *et al.* 1995). As benthic larval assemblages are sedentary in nature, changes in their community structure or population or species levels may well reflect water quality alteration (Rosenberg and Resh 1993, Norris and Norris 1995).

However, there has been very limited use made of benthic macroinvertebrate score and index systems in assessing water quality in Southeast Asia. These countries are under rapid economic growth and suffer much from water pollution. Consequently, development of aquatic pollution monitoring methods is urgently required in this region (Dudgeon *et al.* 1994). Recently, Resh (1995) proposed certain macroinvertebrate taxa metrics (mostly mixed with multiple indices) which might be used for monitoring water quality in newly industrialised countries.

Aims of study

We explore the performance of some current score and index systems and wish to test some of them with the Pong catchment water condition, northeast Thailand. These scores and indices are compared between two different sampling methods, the qualitative and quantitative samples. The main aim here is to test whether the qualitative will be reliably effective when compared to the results derived from the conventional quantitative sampling. The data from two sampling types will also be evaluated via multimetrics, scores and indices systems.

Study site description

The Pong catchment is located on the northeastern plateau of Thailand (Fig.1) between 16°00'-17°15' N and 101°15'-103°15' E. It has an area of 15,190 km^2 and ranges in altitude from 88 to 300 meters above sea level. The Pong system consists of two main tributaries, the Pong and the Cheon. These two rivers receive waters flowing mostly from headwater streams located in the Phetchaboon mountain range. Three large dams that supply water for cultivated lands, industrial zones and communities within the catchment regulate the main channel of the Pong.

Twenty-one sampling stations were established across the Pong floodplain. Site P01-P08 is in the upper Floodplain above Ubolratana dam, while sites P09-P21 are located in the lower catchment area. The land of upper catchment part (P01-P08) is mainly modified for agriculture. The middle catchment part, P09-P13 are located along the Pong river reaches below Ubolratana dam. There is the largest pulp-paper mill factory in Southeast Asia located close to the riverbank and discharges its sewage into the Pong River between site P09 and P10. Sites P14-P18 are located in the lowest part of the catchment. The river receive discharge mainly from city sewage, particulary the river reach from sites P15 onwards. The final sampling site on the Pong river is P18 located above the confluence of the Pong and Chi rivers, approximately 5 km above the Mahasarakam weir. Site P19 to P21 are on the Chi River, which are located in the lowermost part of the catchment region. These sites are affected by substratum rock salt intrusion that becomes severe during summer.

Materials and methods

Qualitative versus quantitative sampling

Qualitative samples were collected from edge waters of twenty one sampling stations in February 1996 using a pond net (0.25x0.25 m with 500 µm mesh). The method was standardised by using fifteen-minute sampling on each riverbank at a site. Six quantitative samples were randomly taken from the riverbed at each site. Sampling benthic fauna at upstream reaches used a Surber sampler (0.30 m x 0.30 m with 500 µ m mesh aperture), while sampling in deeper waters downstream necessitated use of an Ekman Grab. Specimens were identified to the lowest taxonomic level possible, censused and preserved in 70% ethanol. These two data sets will be tested through application of multimetrics, scores and diversity indices.

Major water quality parameters were sampled at the same time as the macroinvertebrate sampling. These were water electrical conductivity (EC), total suspension solid (Total SS), phosphate (PO_4), nitrate (NO_3), dissolved oxygen (DO) and biochemical oxygen demand (BOD-5day). The methods used to sample and analyze all water quality variables followed the standard methods described in GEMS/WATER (UNEP, WHO, UNESCO and WMO (1992) and APHA (1992).

The score system

Three score systems are utilised: the BMWP/ASPT (Armitage *et al.* 1983), SIGNAL (Chessman 1995), and Hilsenhoff Biotic Index (Hilsenhoff 1988). Within the first two systems, each invertebrate family is assigned the score created by both methods, and all scores are summed and later divided by the number of families in a sample. The last score system used by this study here is modified slightly from the original Hilsenhoff's which used only 100 organisms caught as the divider while in this study we used the total number of insect individual caught instead.

The diversity indices system

The diversity indices system applied here selects some most commonly used indices in water quality assessment. These are Simpson's, Margalef's, Shannon's and Hurlbert's PIE. All are shown as follows:

(a) Simpson' s Index = $\sum n_i (n_i-1)/n (n-1)$

(b) Margalef's index = $S-1/\log_e n$

(c) Shannon Wiener index = $-\sum (n_i/n \log_e n_i/n)$

(d) Hurlbert's PIE = $(n/n-1)\{(1-\sum (n_i/n)^2\}$

Where
S=the number of species in a sample
n=the number of individuals in a sample
n_i=the number of individuals of a species *i* in a sample

Testing the score and indices systems

At each site the six replicates from quantitative sampling were aggregated as one sample. Similarly, the specimens collected qualitatively from both riverbanks were also combined as one sample. These two data sets will be analysed by various fundamental indices: the richness measures (the so-called multimetrics), these are number of individuals, species richness, family richness and EPT taxa richness. Secondly, the score and indices systems were used. All these results will be compared between quantitative and qualitative sampling.

The score and indices results from both sampling methods were correlated to major water quality variables using Pearson-product moment. This was done as to use the water quality variables as the reference frame of comparison between score and indices systems produced from the two sampling methods. All data were transformed to log (x+1) when necessary to improve normality prior to statistical analyses.

Results

Multimetric approach

Benthic animal abundance

The specimens collected by the two methods were different in total number(t_{20}=6.00, P<0.001) and taxonomic identity. The six replicates taken by the Ekman grab had a combined total of 4215 specimens while the edge sampling yielded 1208 specimens. The grab samples were composed mainly of Diptera (58.5%), Trichoptera (15.8%) and Ephemeroptera (9.3%), whereas the edge samples were dominated by Hemiptera (43.2%), Ephemeroptera (19.4%) and Odonata (16.2%).

Taxa richness

Although the cumulative number of taxa recorded by edge sampling (n=60) was similar to the streambed grab sampling (n=53, t_{20}=2.05, P=0.054), the composition differed greatly. The edge samples mainly consisted of Hemiptera and Odonata whereas grab samples yielded a high relative abundance of Trichoptera, Ephemeroptera and Odonata (Table 1).

The mean number of taxa from edge samples did not differ over the 3 broad regions in the catchment profile (upper mean taxa richness ± SD: 12.3±2.3, middle 10.6±3.0 and lower 10.7±3.5 respectively, $F_{2,18}$=0.810, P<0.46). In contrast, the benthic taxa richness recovered by grab sampling differed over the catchment landscape ($F_{2,18}$=7.96, P<0.003), Fig.2a. The upper catchment had greater richness (13.5±7.2), than the middle and lower catchment 7.4±2.6 and 4.1±1.5 taxa respectively.

Overall at the whole catchment scale, family richness estimates from edge-water and streambed samples were very similar (31 and 34 families respectively).

Benthic families taken by the grab method varied between catchment regions ($F_{2,18}$= 8.237, P=0.003). Streambed samples from the upper region contained more families (12.3±5.9) than those from sites in the middle (7.4±2.6) and lower catchment (4.1± 1.6). The edge-water samples, on the other hand, did not reveal any significant difference in family diversity over the catchment profile. The upper, middle and lower catchment sites had 8.1±2.3, 7.6±2.9, 7.3±3.4 families respectively ($F_{2,18}$=0.183, P=0.83).
The obvious difference between the results derived from two sampling methods is taxa richness. Families with the greatest number of individuals taken by edge sampling were Gerridae (21% of all specimens), Baetidae (17%), Corixidae (16%) and Protoneuridae (13%) while grab sampling yielded large numbers of Chironomidae (47%), Hydropsychidae (13%) and Corbiculidae (6%). More specimens were present in edge-water samples from middle and downstream sites than upstream, whereas the streambed grab samples yielded most individuals in upstream sites. The fundamental metrics, species and family richness of the two different sampling methods are quite different from each other.

Correlation between taxa richness and water quality variables

There was no significant correlation between species richness from edge-water samples and major water quality parameters. The species richness from quantitative data revealed significant negative relation with PO_4, NO_3 and positive with DO (Table 2). The most polluted site was P16. Its average BOD rose to 9.3 mg/L, nutrient levels were also high with NO_3 2.0 mg/L and P 1.4 mg/L. Only 5 Chironomid larvae were found from six-replicate riverbed samples, while eight species of macroinvertebrates were in from edge-water. There were the mayfly Baetis sp., the water bug *Ctenipocoris* sp., *Mesovelia* sp., *Micronecta* sp. *Halobates* sp., damselflies *Ischnura* sp. and *Pseudagion* sp. And the freshwater shrimp *Macrobrachium lanchesteri*. The DO level in site P16 was considerably high, averaging 7.8 mg/L, and resulted from the photosynthesis of unicellular algae *Microcystis*. In this aspect, the high DO level did not always necessarily correspond to benthic taxa abundance or diversity. Four water quality parameters were related to the number of families recovered by quantitative samples, but there was no significant correlation found from qualitative data (Table 3).

Ephemeroptera, Plecoptera and Trichoptera (EPT) index

Twenty three species of EPT were recorded from streambed samples, whereas only 9 species were recovered. The qualitative data had 6 EPT families, while a total of 15 families were recover from riverbed samples (Table 4). Only two major water quality parameters, BOD and DO, showed a significant relationship to EPT taxa richness in streambeds samples (r=0.77, P=0.001 and r=0.23, P=0.108 respectively). There was no significant correlation between EPT taxa richness of both sampling methods and other major water quality pollution parameters, such as conductivity, nitrate, phosphate and total suspension solid. Percent EPT composition (% EPT individuals/Total abundance), relates to major water quality parameters. Percent EPT taxa composition from quantitative samples shows association with BOD (r=0.49, P=0.012), conductivity (r=0.53,P=0.007) and PO4 (r=0.45,P=0.021), but not with TSS and NO_3. There was no significant correlation between percent EPT taxa from qualitive samples and any water quality parameters. EPT family richness from quantitative samples was also positive correlated to DO (r=0.42,P=0.028), and negatively related to BOD (r=-0.39,P=0.042). There was no correlation between quantitative EPT richness and TSS, NO_3 and PO_4. The EPT family richness from qualitative data did not significantly relate to any water quality parameters.

The scoring systems

All biotic scores from quantitative samples tended to agree with trends in water pollution, they showed different correlation values with pollution parameters (Table 5). Only BMWP/ASPT and SIGNAL scores were significantly negatively related to organic pollute (BOD). BMWP/ASPT showed positive correlation to DO. Hilsenhoff's score showed high association with NO3 and PO4 levels. All three score systems were well related to PO_4 levels. All of three scores failed to detect inorganic pollutants (EC and TSS). The scores from qualitative data did not obviously relate to spatial water quality variation (Table 5).

Discussion

The qualitative and quantitative methods made very different results. The results are clearly suggested that the qualitative method cannot replace the conventional quantitative method in the Pong catchment. The use of a pond net to collect macroinvertebrates fauna was shown to be effective and recommended elsewhere (Furse *et al.* 1981, De Pauw and Vanhooren 1983, Wright et al. 1988, Hellawell 1986, Lenat 1988, Abel 1989). This sampling method can be an integral component of rapid bioassessment (Resh and Jackson 1993, Resh 1995). However, the results from this study show very different species composition from the two different sampling methods. So, for reliable results, the quantitative sampling method is recommend in the Pong.

Richness measures

Conventional indices using taxa richness measures were effective, but in this study, such measure have to be derived from quantitative data. Almost all-major water pollution variables were correlated these quantitative richness indices. The much polluted the water are, the less number of benthic taxa richness found. When relating macroinvertebrate family richness to water quality parameters, the results were similar to that when using the species richness. So, it is clearly to conclude that family level identification to quantify water pollution in the Pong catchment is satisfactory.

EPT indices

The EPT index that is widely used in North America also revealed significant differences between sampling methods and habitat characteristics. The EPT collected from edge-water samples had fewer sensitive taxa. Most were Baetid mayfly which generally known as a widespread taxon with a wide toleration range to water pollution (Hellawell 1986). In contrast, the EPT species richness data from streambed was strongly negatively correlated to BOD, which is the critical water quality problem currently, encountered in this region. Thus, the EPT taxa richness from qualitative samples did not reflect any variation in water pollution variables, while the EPT richness from riverbed samples did, and can possible save cost and time by not having to include other taxa groups. The study found that the EPT taxa richness also relates to DO level. So, the EPT contributed another advantage as it reflected the healthy aquatic ecosystems. However, high DO content does not always indicate clean waters, because it can result from algal bloom which occurs frequently in the lower reaches of the Pong. The correlation of EPT family richness and percent EPT abundance/Total abundance to water quality in this study are considerably inferior to the EPT species richness measure.

Biotic score systems

All three score systems examined by this study performed similarly in relation to water pollution. Thus, all three scoring methods are well correlated to organic water pollution. Only macroinvertebrate families taken from the riverbed contributed meaningful results in relation to water quality. Data from edge-water showed no significant correlation to any water quality parameters. The BMWP/ASPT score was relatively related to DO, BOD and PO_4. BOD and PO_4 are critical water quality pollutants in this catchment, particularly where the river receives a high quantity of sewage discharges, resulting in very low BMWP/ASPT scores for these river stretches. The BMWP/ASPT is sensitive to organic pollution; this results is broadly the same as reported elsewhere (Murphy 1978, Bargos *et al.* 1990, Rossaro and Pietrangelo 1993). Another advantage of BMWP/ASPT score is that most families listed in BMWP are also found in Thailand. Even the BMWP/ASPT contains benthic macroinvertebrate families' scores base on their tolerance values test in Great Britain, but to some extent this can practically be applied to the tropical Pong river. Anyway, this study found that the low score families listed in BMWP which mean High tolerant also inhabited in very polluted area in this region, while the high-score taxa (less tolerant) were found mostly in minimally impacted sites. Resh and Jackson (1993) recommended that this score might need modification for use in different geographical areas. The study suggests that only slight adjustment may be required to apply this BMWP score system for assessing water quality in the Pong catchment. The SIGNAL score system only significantly related to two variables, BOD and PO_4, and less reliable in detecting the most significant water quality variable DO. Hilsenhoff's had poor performance in detecting organic pollution because it had no significant correlation to BOD. However, this score system has high sensitivity to nutrient levels (NO_3 and PO_4). This system failed to discriminate between impact and less impact sites in the Pong catchment, while both BMWP/ASPT and SIGNAL did.

Diversity indices

Only the data from riverbed showed significant correlation to water quality variables. The Shannon Wiener index showed significant correlates to DO and BOD. Simpson's and Hurlbert's PIE indices both showed significant correlation only with DO. Margalef's indices, on the other hand, did not reveal significant association with any water parameters. So, the Shannon Wiener index performed well in relation to changes in water quality. The H' value was correlated with BMWP (r=0.67). The H' needs species level identification while the BMWP score system requires family level. In this aspect, the BMWP is more advantage than the H' when one wants to monitor water pollution rapidly.

In summary, the assessment of water quality in the Pong catchment by mean of benthic macroinvertebrate data summarised indices and scores are quite promising. The data from quantitative sampling method was reliable in respond to water quality changes. Richness measures, both at species and family level, were superior to other indices and scores in assessing water quality changes. EPT richness measures are also valid in a similar way to taxa richness. The BMWP/ASPT score system was more reliable in detecting organic pollution while other scoring methods performed poorly. Of the four diversity indices tested, the Shannon-Wiener index (H') was more sensitive to water quality variation than the others.

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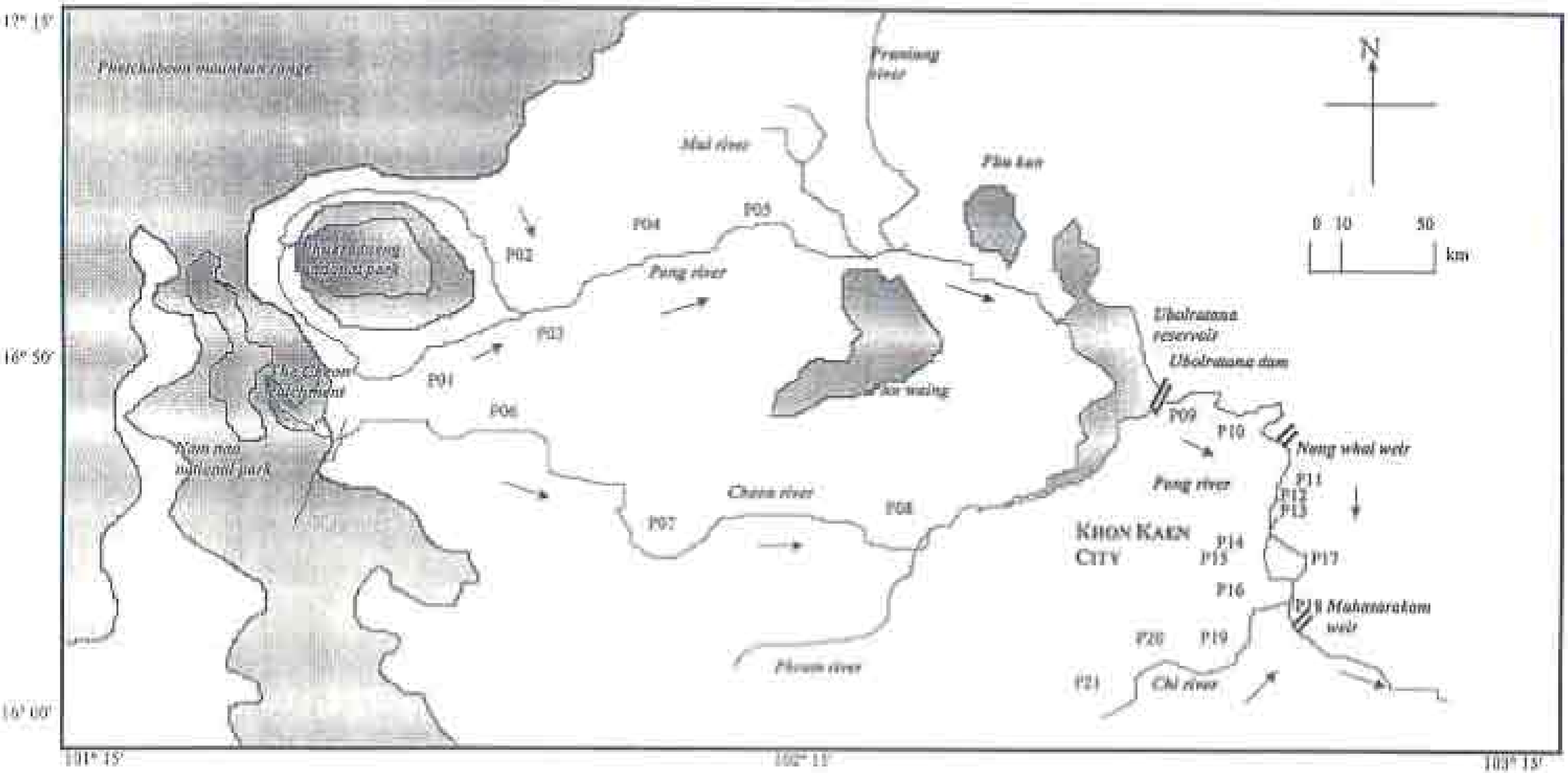

Fig. 1 Sampling sites of the Pong catchment northeast Thailand (Phu means mountain, P01-P08=Upper zone; P09-P21=Lower zone)

Table 1 Species composition between quantitative and qualitative sampling methods

Taxa	Quantitative		Qualitative	
	No. of species	percent	No. of species	percent
Coleoptera	7	13	6	10
Decapoda	1	2	1	2
Diptera	5	9	2	3
Ephemeroptera	10	19	7	12
Hemiptera	5	9	24	40
Odonata	10	19	15	25
Oligochaeta	1	2	0	0
Lepidoptera	0	0	1	2
Plecoptera	1	2	0	0
Trichoptera	12	23	2	3
Veneroida	1	2	2	3
Total	53	100	60	100

Table 2 Correlations between species richness from edge-water and streambed samples and major water quality variables (* indicate significant correlation)

Major water quality variables	Quantitative		Qualitative	
	r	p value	r	p value
EC (micro S/cm)	-0.1782	0.22	-0.2418	0.145
NO_3 (mg/L)	-0.5357*	0.006*	0.0312	0.447
PO_4 (mg/L)	-0.6615*	0.001*	-0.024	0.459
TSS (mg/L)	-0.3173	0.081	-0.0122	0.479
DO (mg/L)	0.4312*	0.011*	-0.1961	0.197
Log BOD (mg/L)	-0.5396*	0.006*	.0.639	0.392

Table 3 Correlations between number of family from edge-water and streambed samples and major water quality variables (* indicate significant correlation)

Major water quality variables	Quantitative		Qualitative	
	r	p value	r	p value
EC (micro S/cm)	-0.1706	0.230	-0.2202	0.169
NO_3 (mg/L)	-0.5710*	0.003*	0.0522	0.411
PO_4 (mg/L)	-0.6807*	0.001*	0.0164	0.472
TSS (mg/L)	-0.3305	0.072	0.0681	0.385
DO (mg/L)	0.4102*	0.022*	-0.1825	0.107
Log BOD (mg/L)	-0.5525	0.005*	0.0463	0.421

Table 4 Comparative percent composition of EPT individuals between quantitative and qualitative samples

Order	Family	Quantitative	Qualitative
Ephemeroptera	Baetidae	9.2	84.8
	Caenidae	16.7	5.5
	Ephemeridae	1.6	0
	Heptageniidae	0.1	0.1
	Leptophlebiidae	6.5	3.4
	Potamantidae	2.7	0
Plecoptera	Perlidae	0.3	0
Trichoptera	Calamoceratidae	0.2	0
	Ecnomidae	3	0
	Hydropsychidae	52.3	0.4
	Hydroptilidae	2.1	0
	Leptoceridae	0.5	1.3
	Polycentropodidae	3.1	0
	Psychomyiidae	0.1	0

Table 5 Pearson-product moment correlations ® between biotic scores and indices and major water quality variables when using (a) quantitative data and (b) qualitative data
(* indicate significant correlation)

(a)

Water quality variables	BMWP's	SIGNAL's	Hilsenhoff's	H'	Hurlbert's PIE	Margalef's	Simpson's
Log EC	-0.3068	-0.0493	-0.0742	0.1659	-0.0828	-0.1038	0.0828
	P=.088	P=.416	P=.375	P=.236	P=.361	P=.327	P=.361
NO_3	-0.227	-0.3576	0.732*	0.0936	0.1259	0.0126	-0.1259
	P=.161	P=.056	P=.0001	P=.343	P=.293	P=.478	P=.293
PO_4	-0.4312*	-0.5787*	0.6308*	0.2593	0.0012	-0.0009	-0.0012
	P=.025	P=.003	P=.001	P=.123	P=.498	P=.498	P=.498
Log TSS	-0.3463	-0.3315	0.3387	0.364	-0.2856	-0.2436	0.2856
	P=.062	P=.071	P=.067	P=.052	P=.105	P=.144	P=.105
DO	0.5574*	0.2777	0.2255	0.4496	0.4209*	-0.202	-0.4209*
	P=.004	P=.111	P=.163	P=.020	P=.029	P=.190	P=.029
Log BOD	-0.4238*	-0.5226*	0.3501	-0.3846*	-0.2768	0.0712	0.2768
	P=.028	P=.008	P=.060	P=.043	P=.112	P=.380	P=.112

(b)

Water quality variables	BMWP's	SIGNAL's	Hilsenhoff's	H'	Hurlbert's PIE	Margalef's	Simpson's
Log EC	-0.1602	0.1359	0.4051	0.0693	0.1446	-0.1828	-0.1446
	P=.488	P=.557	P=.068	P=.765	P=.532	P=.428	P=.532
NO_3	0.235	0.2871	0.1021	0.2483	0.2956	-0.0206	-0.2956
	P=.305	P=.207	P=.660	P=.278	P=.193	P=.929	P=.193
PO_4	0.0095	0.351	0.1373	0.2253	0.2724	-0.0422	-0.2724
	P=.967	P=.119	P=.553	P=.326	P=.232	P=.856	P=.232
LOG TSS	0.0167	-0.1028	-0.2563	0.1476	0.3224	-0.2183	-0.3224
	P=.943	P=.657	P=.262	P=.523	P=.154	P=.342	P=.154
DO	0.3960	0.1086	-0.1173	-0.2745	-0.2243	-0.237	0.2243
	P=.075	P=.639	P=.613	P=.228	P=.328	P=.301	P=.328
Log BOD	-0.0238	0.1918	0.1528	0.3741	0.3949	0.07	-0.3949
	P=.918	P=.405	P=.508	P=.095	P=.076	P=.763	P=.076

Effect of Headwater Catchment Degradation on Water Quality and Benthic Macroinvertebrate Community in Northeast Thailand

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Abstract

Six replicates of benthic macroinvertebrates and stream water were sampling bimonthly for macroinvertebrate fauna composition and water physicochemistry analysis from six sampling sites along the Cheon headwater catchment in northeast Thailand from October 1995 to August 1996. The aims of this study were to examine the catchment water physicochemical quality variation as well as its aquatic benthic macroinvertebrate community and other related aquatic terrestrial environment factors. It was found that degradation of the Cheon headwater was evident by water physicochemical parameters and benthic macroinvertebrates. The water quality in the forest land sites was less degraded than at bared sites. The apparent degradation of water quality in this catchment was caused by clearing for agriculture. The streams covered by forests were less affect by surface runoff, resulting in high DO and low BOD level. The DO, BOD and SS levels resulting from land clearing have influential effects on macroinvertebrates. The macroinvertebrate analysis more accurately reflected these impacts in the streams than the water physicochemistry. The species richness and abundance in the disturbed sites were less diverse than in the pristine sites. The degree of fauna composition in disturbed areas varied according to the extent of land clearing. Among the benthic macroinvertebrate fauna, trichopteran was the most distinctive taxon which was more abundant in very pristine site than the other sites. The net-spinning caddis larvae and riffle bettle *Cleptelmis* sp. clearly showed significance of good forest cover condition and high DO level. While sensitive Heptageniid and Ephemerid mayflies nymphs were found only strictly in microhabitats with high DO level but not strongly related to forest cover or land clearing. Oligochaete, dipteran and mollusc were abundance at impact sites, as they were pollution indicator taxa. Classification and ordination methods identify clear environmental impact influenced by both natural and human causes.

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Introduction

The two main sources of human influence on the most Asian stream and river ecosystems are land modification for crop cultivation and river regulation and control (Dudgeon 1992, 1994b). Tangtam and Aimpan (1995) reported that the Cheon headwater catchment area has a forest-land loss rate as high as 2.1 percent per year and has suffered severely from extensive top soil loss, as high as 18 ton/ha/year. Such excessive runoffs from this catchment are mainly from the absence of vegetation cover since 70% of the land has been cleared for corn and sugar cane cultivation. They also stated that the primary key to relief from such vast sediment flowing into the streams was to restore and manage the riparian vegtation that lined the riparian zone. The Cheon headwater catchment used to be a very important rainfall catchment which perpetually supplied waters to northeast Thailand, but it is no longer contributing waters to the lower flood plain. The Thai government has now initiated a rehabilitation program for the Cheon. However there remains a large number of studies required, generally regarding aquatic environment information (Tangtam and Aimpan 1995). Impacts due to this land modification on floral and faunal communities are still not known. Macroinvertebrate fauna , both at community and species levels, is one of several well-known criteria used to assess land-water related environmental impact (Hellawell 1978, Wright *et al.* 1984, Abel 1989, Plafkin *et al.* 1989, Friedrich, Chapman and Beim 1992, Lenat and Barbour 1994, Barton 1996 and Magati 1996). In the case of catchment studies, distribution of macroinvertebrate fauna is found to be an effective tool in quantifying either stream degradation or restoration (Richards and Minshall 1992, Richards and Host 1993). Most studies of macroinvertebrate communities associated with environmental degradation have been conducted in the temperature zone, but very few in Asia. According to Chaiyarach (1980) and Dudgeon (1994), aquatic environment studies in tropical Asia are inevitably needed in order to contribute to better water resources management. The major objective of this research is to establish baseline data and to study environmental variation of a tropical catchment headwater, the Cheon catchment. Particular emphases are (i) examining the catchment water physicochemical quality variation (ii) corresponding of water physicochemistry to aquatic benthic macroinvertebrates community change (iii) to explore wheather its variation can reflect adverse effects from environmental degradation.

Description of the Cheon catchment

The Cheon catchment is located on the northeastern plateau of Thailand, 570 km northeast of Bangkok and has an area of 727 kilometers. It lies between 16°30'-16°54'N, 101°30'-101°55' E, and its altitude ranges from 280-780 m.s.l. The landscape of the Cheon is mostly hilly valleys and its landform is classified as a slope complex, with slops ranging from 6-27%. Most of the catchment terrain is modified for growing seasonal crops, especially mixed horticultural crops on land slopes which are mostly cleared. The rainfall regime of the catchment differs, with the upper flood plain having a seven-month rainy period from late April to October, while the lower has a wet season of six-month duration from May to October. Its average annual rainfall is about one third higher at 1378 mm whereas the lower flood plain receives 1034 mm. The annual average temperature of the catchment is slightly lower than the lower plain, reaching a maximum in April and May average 29.2 °C, while the coolest month average 20.7 °C in December.

In March, the evaporation rate of the catchment is at its highest, and humidity is at its lowest. It appears as a very dry and harsh landscape. During this time of the year, the bush fires and forest burning by local inhabitant occur frequently. March to early April is the time when the local people are most likely to clear the land for planting crops.

The annual climatic regime of the catchment, in general features a pattern of wet and dry periods. Geophysical structure of the Cheon catchment is mainly sedimentary rock which mostly consists of sandstone, shale and siltstone substrata. The surface soils with an average maximum depth of 0.5 m, are mixed with sandy and clayey loam and belong to the great group-red yellow podsolic soil. With intensive crops growing in this catchment during past decades, the lands then largely became less fertile, and thus extensive fertilisers are applied. High levels of suspended solids diffused from the croplands were observed during pre-surveys by this study. They caused the catchment streams and rivers to appear very turbid, especially during the rainy season. The streams run particularly across a landscape where there is less forest cover, and both coarse and fine solids diffusion dramatically affected their waters.

Although large areas of the catchment land have been modified for massive crop planting, there still remains remnant vegetation within the catchment. Most vegetation patches are standing trees and shrubs which line the river banks. These patchy trees are thus expected to have a substantial role in supporting the aquatic ecosystem function within this catchment. Also, these riparian strips act as barriers for preventing excessive surface runoff from the croplands flowing into the catchment streams.

It can be observed that wherever streams of this catchment have thick strips of buffer vegetation, the water appears to be relatively clear and less turbid, particularly during the storm season. Such riparian vegetation corridors are mostly of mixed vegetation communities, mainly deciduous and dry evergreen trees. The locals seem inclined to preserve these vegetation strips, especially those in the upper catchment. However, at the lower catchment, the locals continue to use the riparian zone for growing crop up to the stream edge.

The only remaining large area of forest is located in the northwest of the Cheon basin (Fig. 1). The vegetation community here is mainly dry evergreen forest and covers approximately 15% of the total catchmant area. This is where the first sampling station was established, and it is intended to be a reference site for comparison with other sampling sites. The forest community of this site is relatively pristine and well protected by the Nam Nao National Park.

Study site description

The sites were established at pristine, moderate and severely disturbed locations, as well as lower in stream reaches. There were six sampling sites in total. The reference site (A) was situated in a protected area of dense forest, located 30 km from Nam Nao District. This site, in particular, was to be compared with other human disturbed sites (B to F in Fig. 1). Site B was about 10 km northeast of site A, but was considerably affected by nearby lands which were heavily modified for cornfields. Site B had a minute riparian strip approximately 1-2 m width. These two sampling sites were located in the second order streams which are tributaries of the Cheon river. Both sites C and D were located in the northeast part of the catchment, and were second order streams. The lands around site C were mostly cleared for cornfields and only some buffer vegetation strips were left along the riparian zone. Site D was located in the stream that runs across a Bhuddist monastery area. The vegetation in this site is relatively protected, and has a dense buffer strip covering the streamside. Site E was located in the southeastern sectors of the catchment, and is characteristic of the lowland Cheon water course. Here the river channel morphology become wider and deeper, particularly during rainy months. A patch of standing vegetation also existed, located sparsely along the river bank. The water current sometimes appeared to be slow and stagnant, particularly during summer. The surrounding lands of this site were cleared for massive sugar cane plantation.

The last sampling site F was located immediately above the Pong lower flood plain. This river stretch was similar to site E, but the adjacent lands were used mainly for residential and agricultural purposes.

Local communities inhabited the riverbanks at site F where they also grew a large variety of vegetables.

Materials and Methods

Sampling was conducted bimonthly at the six sampling sites, from October 1995 to August 1996. Six replicates were applied to sample benthic animals at each sampling site. A uniform stretch of river waterway at each sampling site approximately 100 m in length was marked, and six replicates were sampled randomly within it. Water physicochemistry data were concurrently collected on the same date of benthic fauna sampling.

Sampling water physicochemistry used the van Don bottle at the mid depth of water column where each of the faunal sampling unit were located. The integrated water sample technique was applied by mixing those six water samples in order to represent the water physicochemistry quality at a site. Preservation and analysis of water samples followed the standard methods described in APHA (1992).

Water current and discharge were measured in the field. The water flow-rate and discharge were measured applying the conventional hydrological methods as described in WMO (1980) and discharge values were calculated. All field practices for sampling water physicochemistry followed the guideline methods described in the GEMS/WATER operational guide by UNEP, WHO, UNESCO and WMO (1992). Detailed habitat condition at a site of each visit was also recorded.

Seventeen water physicochemical variables were examined in the field and laboratory. The variables measured in the field were water velocity, stream depth and width, discharge, water temperature, electrical conductivity (EC), dissolved solids (TDS), dissolved oxygen (DO) and pH. Those water physicochemical variables analysed in the laboratory were alkalinity, turbidity, suspended solids (SS), orthophosphate (PO_4), reactive dissolved nitrate-nitrogen (NO_3), biochemical oxygen demand (BOD), chloride (Cl) and sulfate (SO_4).

In this study the *SS* level was used to indirectly quantify the magnitude of surface runoffs, and also the sediments input into streams from surrounding lands. The SS was expected to cause microhabitat alteration detrimental for macroinvertebrate colonisation.

Turbidity was selected to measure the visual properties or transparency of water. The turbidity level was expected to condition the degree of light penetration within a water column. Further, the magnitude of turbidity level would determine the extent of photosynthesis available for benthic faunal community as allochonous food sources.

The TDS and EC were used as rough indicators of mineral salt content in waters, when direct measurement of each dissolved ion could not be made.

Benthic fauna was quantitatively sampled following the general methods recommended by Hellawell (1986). Sampling benthic fauna at shallow upstream sites used a surber sampler (0.30x0.30 m with 500 µm mesh aperture). An Ekman grab (0.15x0.15 m) was used to sample benthic animals in deeper waters at downstream sites.

Benthic samples taken in the field were first collected into polyethylene plastic bags and preserved with 90% ethyl alcohol. The benthic samples were then brought to the laboratory where they were washed and sieved using a series of standard sieves in which the last layer retained was at 500 µm mesh screen.

The animal samples were then hand-sorted on white trays using forceps. All specimens were identified to the lowest possible taxonomic level using available keys. The specimens were further enumerated and lastly preserved with 70% ethyl alcohol. All of the specimens were labelled and contained in vials.

Data analysis

Description

Taxa richness, faunal density, number of organisms, percentage of faunal composition all used descriptive statistics. Faunal variation between time and space employed univariate analysis. Univariate analysis was also applied to the water quality data set. All descriptive and univariate analyses employed the SPSS package (SPSS 1994). All of the data were tested for normal distribution. Log transformation was used whenever necessary as to improve normality. The fauna and environmental data sets were further analysed by multivariate analysis techniques. The software used was PATN (Belbin 1995).

Classification

The sampling sites based on water physicochemical data were clustered by the hierarchical agglomerative clustering UPGMA (Unweighted Pair Group arithMetic Averaging, Sneath and Sokal 1973).

As water physicochemical variables had different scale of measurement, prior to using UPGMA clustering method, they were first standardised by mean/standard devistion. The water physicochemistry data were clustered based on Euclidean distance association metric.

The fauna data were classified using TWINSPAN (Two-way Indicator Species Analysis). Concordance between TWINSPAN classification site groupings and site ordination patterns was also explored. Discriminant function analysis (DFA) was used to find combinations of predetermined significant environmental variables which best predicts the TWINSPAN site groups (Wright 1995). This significance level set for all analyses was 0.05 confidence interval, unless otherwise specified.

Ordination

The faunal data set was ordinated by HMDS (semi-strong Hybrid MultiDimensional Scaling, Belbin 1995), using the Bray-Curtis association metric. Associations between faunal ordination axes and environmental variables were examined using Pearson product-moment correlation. Correlation between ordination axes and macroinvertebrate taxa were sought using the pricipal axis correlation method (PCC option in PATN) to test which taxa significantly correlated to the ordination axes.

Results

Water physicochemistry

Stream discharges within the Cheon catchment varied significantly by site ($F_{2,38}$=8.63, p <0.001). The second order streams had lower water discharges than the fourth and fifth order stream sites. The mean monthly discharge of the catchment reachod a maximum in October, the last rainy month, with an average of 8.52 cu.m/sec. The minimum discharge was in a cooler and dry month in February with its lowest level, 1.32 cu.m/sec. The water discharge level of the catchment sites were also significantly positively correlated to suspended solids (r=0.58, p=0.002), turbidity (r=0.49, p=0.003) and velocity level (r=0.44, p=0.009).

Water velocity, suspended solids and turbidity levels of the catchment streams reflected the temporal and spatial discharge pattern. The average maximum *SS* level, in particular, occurred in October is 171.2 mg/L and the minimum level in February, 3.0 mg/L. The highest *SS* level was mainly resulting from a high amount of surface discharge flux along the catchment waterways. At this time the water became very turbid and many floating plant fragments floated along the river channels, particularly at lower stream reaches. The upstream sites had an average *SS* level lower than the downstream sites, which average were 57.3 mg/L and 74.6 mg/L, respectively. Water velocity varied considerably through seasonal regime, maximum velocity in rainy season was 1.4 m/sec while minimum velocity is in dry season. The water became almost stagnant in some stream reaches. The *SS* and turbidity levels were well related, but they reached their peak in different times. The *SS* showed the peak value in October, averaging 171.3 mg/L, while the turbidity reached the maximum level in August with a mean of 74.5 NTU.

Air temperature varied markedly between sites. The upper site had significantly lower average temperature than the lower sites, which ranged from 24.4-25.5 °C and 27.5-27.6 °C respectively (t_4=5.96, $p<0.01$). Average air temperature of all sampling sites reached a maximum in April, 27.6 °C, while dropping to a minimum in December, 22.4 °C.

The average water temperature significantly differed between sites. The first two upper stream site (A and B) with relatively high percent forest cover had markedly lower water temperature than other sites, which averaged 20.3 °C and 24.2 °C, respectively. The water temperature levels also varied between dry and wet seasons ($F_{1,35}$=61.34, $p<0.001$).

The water temperature rose to its highest level 25.1 °C in April and then decreased to a minimum 17.2 °C in December. The stream sites located in areas with denser forest cover (A and B) had lower water temperature than the more exposed sites.

The nutrient levels of the catchment waters varied significantly by sites and months. Th PO_4 levels ranged from non-detectable levels to a maximum of 1.10 mg/L, with the mean 0.11 mg/L. The NO_3 levels ranged from non-detectable value to the highest level 0.54 mg/L, with the mean 0.15 mg/L. The NO_3 levels varied significantly between months ($F_{5,18}$=3.14, $p<0.05$). The NO_3 in waters were relatively higher in February and April, when they averaged 0.25 and 0.23 mg/L respectively. The lowest level of NO_3 measured 0.02 mg/L was in the minimal discharge month of December. The annual average NO_3 level of the Cheon was 0.15 mg/L.

The PO_4 levels of the Cheon varied significantly between months ($F_{5,28}$=15.21, p<0.001), but not by sites ($F_{5,28}$=0.33, p>0.05). The average PO_4 level of all sampling sites was 0.11 mg/L. In wet season, the average dissolved PO_4 level was higher, 0.14 mg/L while during dry period was 0.04 mg/L. Like NO_3, the PO_4 increased to its peak level in April, the first rainy month, averaging 0.56 mg/L.

The highest average TDS level, 170.84 mg/L, was in February and decreased to an average of 106.96 mg/L in June and October. The site A and B had distinctively lower TDS value than other sampling sites. TDS level varied significantly between months ($F_{5,28}$=3.62, p<0.01). The EC values of all the sampling sites ranged from 62.4 to 541.3 µS/cm, with a mean of 207.3 µS/cm. The EC levels in December and February were relatively higher than in other months. Sites A and B had distinctively lower TDS and EC than other sampling sites. The alkalinity values as $CaCO_3$ in all sampling sites fluctuated similar to TDS and EC levels, and ranged from 36.0 to 318.0 mg/L. The average alkalinity level of the catchment water was 100.6 mg/L. Alkalinity levels did not vary between months ($F_{5,28}$=[illegible], p>0.05), but they were spatially different ($F_{5,28}$=9.16, p<0.01). Site A, B and F had lower alkalinity levels which were 75.3, 52.0 and 85.0 mg/L, respectively. Sites C, D and E, in contrast, had relatively higher alkalinity values, which were 109.7, 110.0 and 174.7 mg/L, respectively.

The pH levels was insignificant between sites ($F_{5,28}$=0.79, p<0.05) and months ($F_{5,28}$=2.31, p <0.05). The pH levels of all sites ranged from 6.2 to 8.8, with a mean of 7.8. The SO_4 levels ranged from 0.5 to 22.1 mg/L, with the mean level 10.3 mg/L. The SO_4 level did not significantly differ between sites ($F_{5,28}$=1.31, p>0.05), but it distinctively varied by months ($F_{5,28}$=6.34, p<0.01). The SO_4 level was higher in wet months than in dry season. Similar to SO_4 value, the Cl. level was relatively low, which range from 3.5 to 15.9 mg/L. The Cl level varied significantly between sites ($F_{5,28}$=7.56, p<0.01), but not by months ($F_{5,28}$=1.33, p>0.05). The highest Cl level occurred in December. The DO level varied significantly between months ($F_{5,28}$=3.51, p<0.05). The lowest level, averaging 4.9 mg/L was in April and the averaging highest level was 7.8 mg/L in December. The DO values were negatively correlated with SS (r= -0.72, p=0.001) and TDS (r= -0.77, p=0.001). The BOD level of the catchment water varied significantly between seasons ($F_{1,33}$=4.35, p<0.05). In the dry season the mean BOD level was 1.7 mg/L, while in the rainy period it was 1.2 mg/L.

Site clustering by water physicochemistry data

Figure 2 shows the dendrogram of sites clustered by UPGMA based on seasonal water physicochemical data. Most of the sampling sites are clearly separated into two main clusters, and largely following seasonal regime (dry and rain). Water samples taken from both dry and wet seasons in site D are still attached to the same cluster, inferring that water quality in site D did not change in either season.

Other sites, apart from site D, are well separated in agreement with samples taken by seasonal regime. The strongest influential physicochemical variables which discriminant the grouped sites were turbidity, SS, PO_4 (Kruskal-Wallis =8.562, df=2, p=0.0138) and SO_4 (Kruskal-Wallis =7.208, df=2, p=0.0272). During the rainy period, these significant variables were higher in levels when compared to the dry season. Site A, the less disturbed site, in particular, is comparatively well segregated from other sites in both seasons.

Benthic macroinvertebrate community and their species variability

Benthic macroinvertebrates representing a total of 13 orders, 57 families, 99 species were discovered. Chironomidae and Oligochaeta, however, were counted as a single taxon each. The most diverse benthic macroinvertebrates are the caddisflies with a total of 23 species, and accounting for 24 % of the total number of benthic species discovered. Mayflies constitute the second largest group, with 17 species. Among insect larval species, the rarest species are lepidopteran and plecopteran taxa with 1 and 2 species each. Dipteran taxa have the highest abundance with its density of 211.15 specimen/m^2. The next most dominant larval taxa are caddisflies and mayflies, with average densities of 178.3 and 118.9 specimen/m^2, respectively. The control site A has the highest benthic animal density with 214.8 specimen/m^2. Site B to D have 81.7, 128.9 and 17.6 specimen/m^2. The downstream sites, E and F, have relatively minimal individual density, which are 7.2 and 17.2 specimen/m^2, respectively.

The benthic density and species richness varied between months. Both benthic species richness and densities increased in the dry-cool months, particularly in December and February. During April, the first rainy month, benthic species richness and density declined gradually until the end of the monsoon season in October. The highest individual density occurred in December, 217.5 organism/m^2, and lowest in October 52.5 organism/m^2. The species richness also followed the density pattern, and was highest in December 13.2 species and lowest in August 8.7 species.

Reference site A, located in a healthy forest, has notably the largest density of trichopterans, 319.16 specimens/m^2, while sites B to C have lower caddis individual density. Gastropod, bivalve and Oligochaeta taxa were absent from site A during the one-year sampling. Oligochaeta, in particular, is abundant in site C, 180.6 specimens/m^2, eventhough this site was located in the same plain as site A. Taxa richness of sites A to F are 41, 19, 57, 48, 16 and 14 taxa, respetively. Taxa richness and density declines downstream. Comparing between upstream (A to D) and downstream (E to F), the upper sites have benthic 96 species with a mean individual density of 128.9 specimen/m^2, while lower sites have 21 specimens and 17.2 individual/m^2.

The less disturbed site A, has the highest macroinvertebrate fauna density, and the maximum number of caddisfly species. The most common in site A were the trichopterans: *Cheumatopsyche malaysiensis*, with 1851.8 organism/m^2, *Synaptopsyche klakahana*, with 1188.8 individual/m^2, and the ephemeropteran *Potamanthus* sp. with 966.7 specimen/m^2. Site B located next to site A, has a total of 19 benthic species identified. Most abundant species were dipteran Chironomidae, the odonatan *Sinogomphus* sp. and the mayfly *Ephemera* sp. with densities of 240.0, 166.7, and 166.7 organism/m^2, respectively. In site C, the most dominant species are the mayflies *Choroterpes* sp., *Ephemera* sp. and the caddisfly *Polycentropus* sp. with densities of 283.3, 233.3 and 183.3 organism/m^2, respectively. Site D, the intermittent stream, is dominanted by mayflies *Ephemera* sp., *Heptagenia* sp. and Chironomidae with 438.9, 355.6 and 375.0 individual/m^2, respectively. The two lowland sites, E and F, are dominanted by dipterans, Oligochaeta and bivalves. The individual densities of Oligochaeta and bivalve *Corbicula brandiana* which dominate site E are 142.6 and 131.7 organism/m^2. The dominant species of site F are Chironomidae and Oligochaeta, which have the same density of 38.9 organism/m^2.

Multivariate analyses of benthic community data

The result of TWINSPAN classification using faunal density data is shown in Figure 3. Eight sample groupings are split at the third level of TWINSPAN division. The samples collected from site E and F are separated at the first division. However, one of the upstream samples, B6, is included in this group. The indicator species contributing to the split on the positive side is Oligochaeta, while on the negative side is mayfly *Caenis* sp1.

On the negative side: at level 2, all upstream samples are split into four groups, group 1 to 4. Indicator species at level 2 is ephemeropteran *Habrophlebiodes* sp. At level 3, indicator species are the elmid beetle *Cleptelmis* sp., Oligocheata, the ceratopogonid dipteran *Bezzia* sp. and the megalopteran *Sialis* sp.

On the positive side: at level 2, indicator species are the dipteran *Bezzia* sp., bivalve *Corbicula brandiana*, Oligochaeta and elmid coleopteran *Stenelmis* sp. At level 3, the indicator taxa are trichopteran *Phylocentropus* sp. and coleopteran *Stenelmis* sp.

Sample collected from downstream sites are clearly separated into one major group at level 1 (group 5 to 8), eventhough these samples were collected in different months. The indicators species produced from the TWINSPAN agree with the samples/sites finding. The profundal ceratopogonid *Bezzia* sp., nematode Oligochaeta and bivalve *Corbicula brandiana* all prefer to inhibit downstream sites.

The occurrence of the biting midge *Bezzia* sp. was related to water pollution. *Bezzia* was often found in upstream site C. For example, in April, the first monsoon month, site c had the greatest *Bezzia* sp. density 255.6 organism/m^2 which was the highest density of this species in all sites. At this time, site C also had high average BOD, EC and TDS level, which were 4.5 mg/L, 313.5S/cm and 208.5 mg/L, respectively.

Like *Bezzia* sp., Oligochaeta is an indicator taxon that can identify polluted water samples, particularly the fine separation of samples between group 3 and 4. The contrast between samples taken from sites C and A is apparent: site C had high average Oligochaeta density, 466.7 organism/m^2 while there was no Oligochaeta in site A.

The megalopteran *Sialis* sp. is a significant indicator species that discriminates between samples within group 4. This species was found in two sites, B and C. Site C had a maximum abundance of *Sialis* in December of 133.3 organism/m^2.

Water quality variables differed markedly between TWINSPAN groups, particularly BOD, SS and turbidity ($F_{7,28}$=3.439, $p<0.05$, $F_{7,28}$=7.187, $p<0.05$, $F_{7,28}$=3.218, $p<0.05$ respectively). TWINSPAN group 3 and 4 had relatively cleaner water quality. Also, both groups had more species richness than the other groups. The finer difference between sample groups 3 and 4 was indicated by the presence of riffle bettle *Cleptelmis* sp. in group 3. Waters in group 3 had comparatively low BOD, SS and turbidity levels, which were 1.1 mg/L, 19.2 mg/L and 17.1 NTU, respectively.

Of all TWINSPAN groups, sample groups 1 to 4 had cleaner water quality and higher species richness than group 5 to 8. Water discharge and SS were identified by DFA to be the most significant variables. The DFA can predict and separate the TWINSPAN groups 1 to 4 clearly, with 100 percent success (Fig. 4). The water quality variables of sample group 3 indicated less impacted than group 4, and the difference between these groups was also reflected in the benthic fauna.

TWINSPAN group 3 was mostly comprised of sensitive species less tolerant to environmental stress, including those listed in Hellawell (1986) and Lenat (1993). These are the trichopterans:*Goera* sp., *Hydroptila* sp., *Tinodes* sp., *Molanna* sp., *Anisocentropus* sp., *Chimarra* sp., *Oxyethira* sp. and *Triaenodes* sp.; the coleopterans: *Cleptelmis* sp. and *Dineutus* sp.; the ephemeropterans: *Thraulodes* sp., *Ephemera* sp., *Paraleptophlebia* sp. and *Heptagenia* sp. and the dipteran *Simulium* sp.

TWINSPAN group 4 had the highest species richness (Table 1) it had less abundant sensitive species than group 3. Sensitive species of group 4 are the trichopterans: *Chimarra* sp., *Triaenodes* sp. and *Ecnomus* sp.; the ephemerans: *Choroterpes* sp. and *Potamanthus* sp.; and the coleopteran *Stenelmis* sp.

Ordination results

The sample ordination results generally resemble those produced by the TWINSPAN. The samples from less impacted sites are ordered at the positive end of ordination axis 1 (Fig. 5a). This group of samples corresponds with TWINSPAN group 3. The samples from impacted sites (B, E and F) are distinctly separated from the rest in the ordination space.

The ordination result can also idenify the sample with more diverse species. Sample A3, in particular, located at the highest positive value on axis 1, had the highest species richness (21 species) suggesting that axis 1 may be a richness vector.

Figure 5b shows the species vectors that highly correlate with the ordination space, as derived from the Monte Carlo test. There are strong agreements between TWINSPAN indicator species (Fig. 3) and HMDS species vectors (Fig. 4). The two most important indicator species produced by TWINSPAN, *Caenis* sp1 and Oligochaeta, are also clearly identified as influential by the HMDS.

Caenis sp1 is the indicator species of the first TWINSPAN division and is identified by the HMDS to have a high correlation with the ordination space (r=0.72). The *Caenis* sp1 vector points towards sites/samples A, C and D, at the middle of the plot. *Caenis* sp1 was a common species that occurred in upstream sites. Oligocheata also highly corelates with the ordinatin space (r=0.81), and its vector points to downstream samples/sites.

The elmid *Cleptelmis* sp. vector increases in the direction of samples of site A and D, where this species was abundant. It was evident that the riffle bettle *Cleptelmis* sp. was abundant only in clear and clean waters, as in TWINSPAN group 3. Thus, the *Cleptelmis* sp. vector identified by HMDS is also confirmed by the indicator species produced by TWINSPAN.

Othert species which highly correlate with the odination space are the mayfly *Ephemera* sp. and the alderfly *Sialis* sp. *Ephemera* sp. was abundant in site A and D, while *Sialis* sp. was often found in site B. All these species vectors point to their corresponding sampling samples/sites in the ordination space. *Sialis* sp. was often found in site B. All these species vectors point to their corresponding samples/sites in the ordination space. *Sialis* sp. is also the indicator species which TWINSPAN used to split the sample groups 3 and 4. *Ephemera* sp., on the otherhand, is not identified as influential by the TWINSPAN. Pearson product-moment correlation test showed that altitude (r=0.78, p<0.001), land use category (r=0.71, p<0.001), buffer strip width (r=-0.67, p<0.001), stream depth (r=-0.71, p<0.001), stream width (r=-0.65, p<0.001) and water physicochemistry: water discharge (F=-0.68, p<0.001), turbidity (r=-0.40, p<0.01), SS (r=-0.69, p<0.001) and water temperature (r=-0.44, p<0.01) are all significantly correlated to axis 1. Buffer strip (r=0.45, p<0.01) and TDS (r=-0.37, p<0.05) are correlated to axis 2.

Discussion ans conclusion

Enviromental impact: water quality variation

The variation of water quality in the Cheon headwater catchment generally followed the monsoonal cycle. Like many tropical Asian streams during the rainy season, there were large amounts of sediment diffused from the adjacent land surface into streams (Dudgeon 1995). The high-low flow regime dramatically altered the stream conditions and benthic faunal habitats, reflected in quite a marked contrast of water quality between wet and dry seasons. In reference site A which located in the protected forest area of Nam Nao National Park, the water quality did not significantly vary, whereas the sites in bare lands critically suffered from land runoffs.

The impact of land use was clear when comparing between sites A to D which were all located at the same altitude and belong to the same stream order. The water quality in the forest land site was less impacted than at exposed sites. SS was the most significant water impurity in this catchment. The sites with thick vegetation strips had relatively lower SS levels than the sites with less riparian zone.

The ambient and water temperature were also influenced by the magnitude of the surrounding forest. The air and water temperatures at well vegetated riparian sites were lower than at the bare sites.

The most serious dissolved nutrient problem of the Cheon was due to PO_4, which fluctuated between seasons. It was very high in the wet season (0.14 mg/L), and minimal in the dry season (0.04 mg/L). The NO_3, however, did not vary by seasonal regime, its levels were 0.16 mg/L and 0.13 mg/L in wet and dy seasons, respectively. In this instance, the results suggested that the NO_3 could be transformed into other forms under the nitrogen cycle. The seasonal rainfall in April was a major cause off diffused nutrient bound sediment from agricultural lands entering into the stream.

Average TDS, EC Chloride and SO_4 level of the Cheon water was relatively low and did not indicate much impact. The levels of pH and alkalinity were relatively high but the values still fall within acceptable range of natural waters. The alkalinity sourse of the Cheon was mainly bicarbonate ion.

Largely, the water samples collected from the Cheon had DO and BOD level that met the national standard value of 6.0 and 1.5 mg/L, respectively (Ministry of Science, Technology and Environment-MSTE 1992). But when considering DO and BOD values between stream sites, these two variables had shown clear water quality impacts. The waters in impacted sites had relatively lower DO content and higher BOD than the less disturbed sites.

The fluctuation of DO values between sites was closely correlated with two combined factors; seasonal regime and SS. During heavy rainfall, the stream sites located in the cleared lands were much impacted from surface runoffs, thus yielding very high SS levels. The diffused sediment further caused the DO depletion within the water column. Like Do values, the BOD of the Cheon water varied between sites. The impacted sites from land clearing had high BOD levels. Diffused sediment from the land surface is the major cause of high BOD within the water column.

The result from UPGMA identified that SS and PO_4 were the two most significant water quality impurities of the Cheon waters. Both values were derived from diffused sediment, and more intensive agriculture, particularly in the stream sites located in cleared lands. The magnitude of water quality impact in the Cheon was greatest during the first rainy month, April.

Regarding water physicochemistry condition, it can be summarised that the Cheon headwater streams showed appearent water quality impacts from land use. The streams located in the bare lands received much impact from diffused solids, which led to high BOD and low DO levels. The waters in stream situated in forestland were less impacted from surface runoffs, resulting in high DO and low BOD. These two water quality parameters are in fact, important determinants which influence aquatic animal life. The benthic macroinvertebrates are animals which are sensitive to water quality changes. The DO, BOD and SS levels resulting from land clearing, have influential effects on macroinvertebrates.

Environmental impact: the presence of indicator species

The remarkedable appearant between the control and impact sites is that the reference site has a large number of species different from the disturbed sites. Disturbed sites with different degrees of impact also had different species composition.

Both sensitive and tolerant species were restricted to different sites and times. Among the macroinvertebrate fauna, Trichoptera was the most distinctive taxon which was more abundant in site A: *Amphipsyche meridiana*, *Cheumatopsyche malaysiensis*, *Tinodes* sp., *Stenopsyche siamensis*, *Hydropsyche* sp., *Neureclipsis* sp. and *Synaptopsyche klukahana*. Most of these species were found during December, the post-flooding period with a rather cool climate. These caddisfly species built their fixed retreat on cobbles. The waters in site A during December were very clean with a high DO level of 8.3 mg/L and a moderate current speed of 0.9 m/sec. In this instance, it suggested that the presence of these caddisfly species was associated with seasonal regime, DO and forest cover.

Some caddis species in site A showed a periodic occurrence following seasonal regime. *A. meridiana* was the first species which appeared only after the post-flooding period. The emergence of *A. meridiana* was also reported in recovery streams after floods in Indonesia (Boon 1984). Similarly the hydropsychid *C. malaysiensis* and *Hydropsyche* sp. were also found in site A after the rainy season. These three filter-feeder caddis built their fixed retreat on submerge cobbles in site A. This site had relatively clean water. These caddis species were classed as mildly to moderately tolerant species by Roux et al. (1992). *A. meridiana* and *C. malaysiensis* inhabited in Yakruea and Phromlaeng stream of Nam Nao National Park, 40 km from the Cheon, with also relatively clean water (Sangpradub and Naknan 1997). It suggested that the presence of these species was still related to mild perturbation.

Another distinctive caddisfly in site A was the polycentropid *Neureclipsis* sp. This net-spinning caddis is also sensitive to organic pollution (Hellawell 1986). Tinodes sp. and *S. siamensis* were found only in site A. Both caddis specis preferred to inhabit in cool, shaded stream with high DO level in site A. These two species were rare in this catchment, but the first species was abundant in pristine streams in Phukadueng National Park, 50 km from the Cheon (Inmuong 1997). Both caddis species were found in Yakruea and Phromlaeng stream of Nam Nao National Park. *S. siamensis* seemed to be resticted to Phromlaeng stream (Sangpradub and Naknan 1997).

Although almost no studies to date associate any single caddisfly species with environmental factors in the Asian tropics, the restriction of these net-spinner species to site A may indicate pristine water conditions. Further, the existence of these caddisflies clearly showed the significance of forest cover to the streams and adjacent lands. Consequently, studies into the influence of land clearing on stream waters might use the presence of net-spinner caddisflies as a biological indicator.

Mayflies also showed remarkedably different distributions between sites. *Potamanthus* sp. was particularly abundant in site A. The BMWP score (Hellawell 1986) assigned a score of 10 (highly sensitive) to all *Potamanthidae* and Lenat (1993) classed *Potamanthus* sp. as a sensitive species to water pollution. This species was very common sprawling on the stream bottom in site A, but rarely found in other sites. *Heptagenia* sp. and *Ephemera* sp. which were found in various sampling occasions in site A to D. These species were found only in streams where the waters had a high DO level. These two mayfly taxa are widespread and recognised elsewhere assensitive species which require highly dissolved oxygen (Hilsenhoff 1988).

Unlike caddisflies, the distribution of these two mayflies was not strongly related to forest cover or land clearing, but rather reflected the dispersion of microhabitats with high DO levels within the streams. This suggests that, within a modified stream reach, there may be still certain microhabitats that could be occupied by these high oxygen-requiring larvae.

Only elmid beetle species, *Cleptelmis* sp. was related to the degree of impact between sites. This beetle family was classed as moderate tolerant species in Great Britain. But Jach and Kidada (1995) noted that most elmids in Asia could be used to indicate good water quality. This is reinforced by this study since the riffle beetle *Cleptelmis* was more abundant in high oxygenated waters both in riffle and run areas in site A. It was mostly limited to site A, and like net-spinnig caddis, is a member of the indicator fauna of high DO and good forest cover condition.

Stoneflies are another taxon commonly used to inicate good water quality (Baumann 1979, Harper 1994). Plecopteran were very rare in this catchment, and only two species of Perlidae were found. *Neoperla sp.* is widespread (Sivec 1984) while *Phanoperla sp.* is abundant in tropical Asia (Harper 1994). Perlidae has been claimed to be a family sensitive to pollution (Hilsenhoff 1988, Stewart 1992). These stonefly species were found in both less impact site A and impacted downstream site E. So, it was less reliable to be as indicator species for water pollution in this catchment. Uchida (1990) and Harper (1994) reported that Asian perlidae occupied both highland and lowland streams and were well adapted to environmental changes.

In contrast to the sensitive species approch, the presence of tolerant species at a site was found to be another valuable indicator capable of differentiating degrees of impact. This study confirmed the status of Oligochaeta, Ceratopogonidae and Chironomidae as pollution indicator taxa, as all three groups were abundant at impacted stations.

Environmental impact: the implication of macroinvertebrate density

Quantifying freshwater catchment impacts using the presence of indicator species was quite a promising approch as discussed above. The density data, in addition, allowed more rigorous analysis in the cool month of December, while the impacted sites had the greatest individual density in the first rainy month of April. The less impacted sites were dominated by caddisfly and mayfly species, while the impacted sites were dominated by dipterans, Oligochaeta and molluscs.

Macroinvertebrate individual density fluctuated markedly following the seasonal climate, largely due to the flooding regime. Within a site, the variation of individual density through time showed a clear relationship with physical water properties, especially discharge, SS and turbidity. Between sites, differences in individual densities were associated with land use and the nature of the riparian buffer strip. The dominant cause that dramatically reduced the individual density was land clearing which led to the elevated water discharge and high SS and turbidity within water column. Finally, the habitat was altered and became unsuitable for macroinvertebrate colonisation as previously found by Dudgeon (1994). Generally, the total individual density data from multiple habitat sampling can reflect the magnitude of impacts on sites. More precisely, a high individual density of certain taxa, particularly caddisfly and mayfly, can indicate more pristine water at a site, whereas high individual densities of Chironomidae, Ceratopogonidae and Oligochaeta, on the other hand, signify more degraded stream sites.

Classification and ordination results

The indicator species produced by the TWINSPAN classification also confirmed the outcomes when using individual density data as discussed above. Both pollution tolerant and sensitive species indicated by the TWINSPAN showed a clear relationship with the magnitude of special and temporal impacts at a site.

Indicator species from the TWINSPAN provide invaluable information as these species strongly relate to the environmental impacts between samples/sites. For example, Figure 3 illustrates the classification results from all samples combined from all months and sites, and it is very difficult to separate samples within groups 3 and 4 if using species richness and individual density data.

The existence of indicator species, for example, *Bezzia* sp., led to the discrimination between samples groups 3 and 4. The presence of the key stone species, *Bezzia* sp., signifies the environmental impact occurring in samples/sites, as in the case of site C which has *Bezzia* sp. abundantly present in February when the water were severely polluted with maximum levels of BOD, EC and TDS.

The impact of discharge and SS from overland flow on species composition in each TWINSPAN group is also obvious. For instance, samples of group 3, in particular, had relatively less polluted waters and had high sensitive species richness, such as trichopterans:*Goera* sp., *Hydroptila* sp., *Tinodes* sp., *Molanna* sp., *Anisocentropus* sp., *Chimarra* sp., *Oxyethira* sp., *Triaenodes* sp., the coleopterans: *Cleptelmis* sp., *Dineutus* sp.; the ephemeropterans; *Thraulodes* sp., *Ephemera* sp., *Paraleptophlebia* sp., *Heptagenia* sp., and the dipteran: *Simulium* sp.

The most useful results produced by TWINSPAN are the indicator species since these provide ecologically meaningful data regarding environmental stress. Both "sensitive" or "pollution" indicator species can be obtained indirectly from the TWINSPAN output while tolerant scores for all species from ecotoxicological test are not yet available.

TWINSPAN effectively classified the less impacted and impacted samples/sites. The ordination results also conformed to the TWINSPAN output. Identifying those samples/sites which were impacted, as well as the indicator species produced by the two methods. This similarity in output suggests these data sets may be used interchangeably in multivariate analyses.

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Figure 2 Dendrogram from hierarchical agglomerative clustering of all sites seasonally sampling sites based on physicochemical variables

Figure 1 Location of sampling sites a-E in the Cheon headwater catchment, with local names in parentheses (redrawn from Tangtam and Aimpan 1995)

Figure 3 Classification of Cheon catchment samples by TWINSPAN. Groupings are retained at the 3rd level of division, including indicator species.

(letters represent site codes, numbers stand for sampling months, 1=October 1995, 2=December 1995, 3=February 1996, 4=April 21996, 5=June 1996 and 6=August 1996)

Figure 4 A biplot between discriminant function 1 and 2, the legend groups corresponding to the TWINSPAN groups in Figure 3.

(Functions 1 and 2 are significant when tested by Wilks' lambda, $p<0.001$, $p<0.05$, respectively)

Figure 5 Biplots between axes 1 and 2 of (a) samples ordination by HMDS (stress 0.1623) and (b) significant species vectors which strongly correlate to ordination space.

(Figure 5a, letters represent site codes, numbers stand for sampling months, 1=October 1995, 2=December 1995, 3=February 1996, 4=April 1996, 5=June 1996 and 6=August 1996.

Figure 5b, Atherix=*Atherix* sp., Caenis1=*Caenis* sp1, Chironom=Chironomidae, Chorotер=*Choroterpes* sp., Cleptelm=*Cleptelmis* sp., Euxomus=*Euxomus* sp., Erpetogom=*Erpetogomphus* sp., Ephemera=*Ephemera* sp., Oligocha=Oligochaeta and Sialis=*Sialis* sp.

Table 1 Number of species per TWINSPAN group

Taxa	TWINSPAN group							
	1	2	3	4	5	6	7	8
Insect taxa								
Coleoptera	0	5	5	10	0	3	1	2
Diptera	0	3	6	6	2	3	2	3
Ephemeroptera	0	7	10	10	2	3	2	0
Hemiptera	0	1	3	4	4	0	0	0
Lepidoptera	0	1	0	0	0	0	0	0
Megaloptera	1	1	1	1	0	0	0	0
Odonata	0	6	3	9	0	1	0	0
Plecoptera	0	0	2	0	2	0	0	0
Trichoptera	0	7	18	14	1	0	0	0
Non-insect taxa								
Mesogastropoda	0	1	0	0	0	0	0	1
Oligochaeta	0	0	0	1	1	1	1	1
Veneroida	0	0	0	1	0	1	1	1
Taxa richness	**1**	**33**	**49**	**57**	**9**	**12**	**8**	**8**
Average density (organisms/m^2)	**77.7**	**224.1**	**104.2**	**123.8**	**68.3**	**10.3**	**89.1**	**45.5**

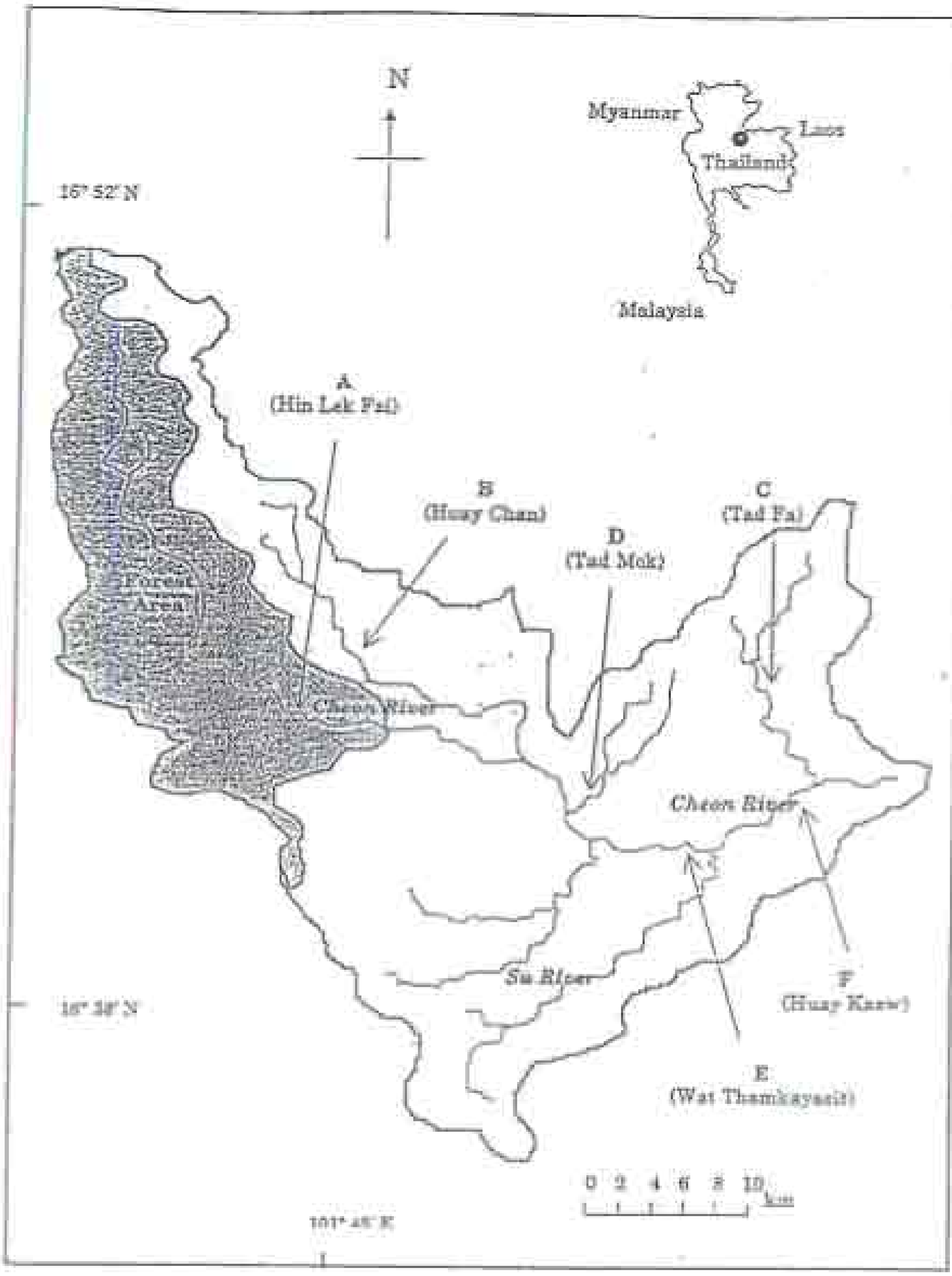

Figure 1 Location of sampling sites a-E in the Cheon headwater catchment, with local names in parentheses (redrawn from Tangtam and Aimpan 1995)

(a)

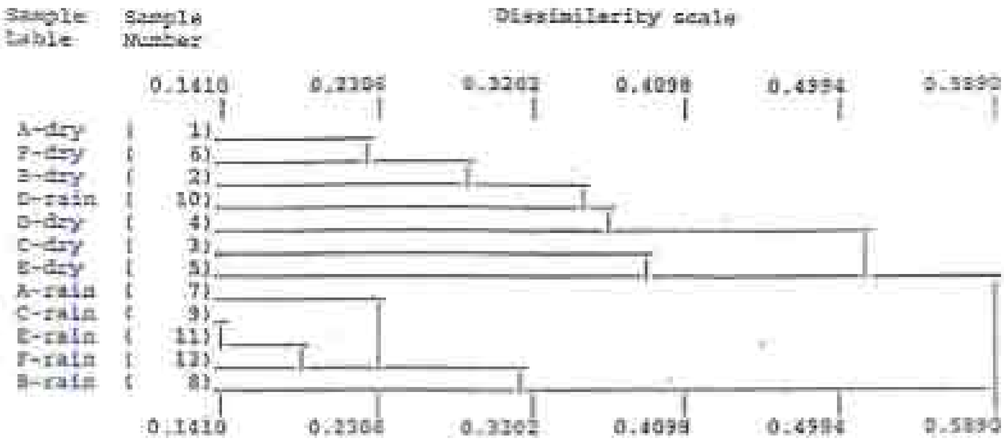

Figure 2 Dendrogram from hierarchical agglomerative clustering of all sites seasonally sampling sites based on physicochemical variables

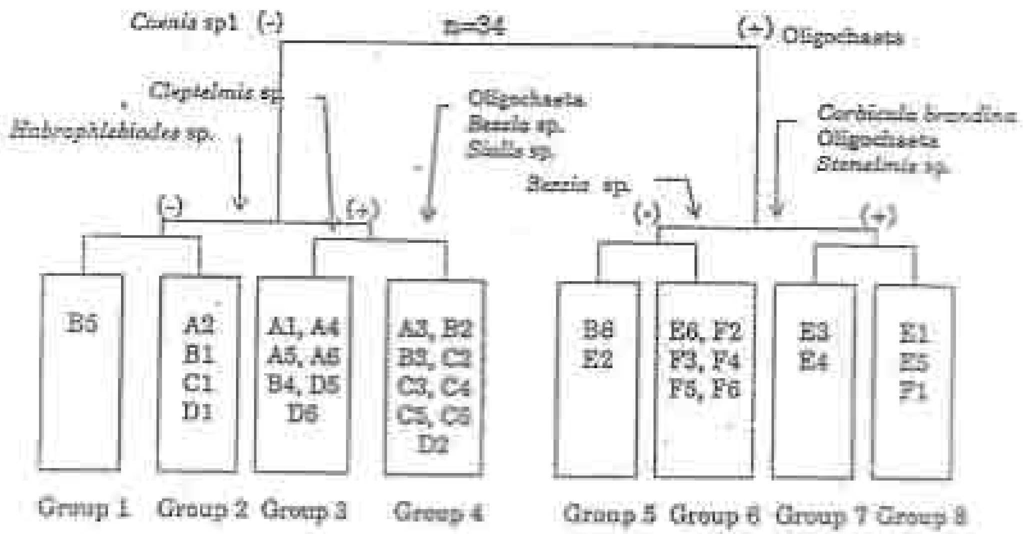

Figure 3 Classification of Cheon catchment samples by TWINSPAN. Groupings are retained at the 3rd level of division, including indicator species.

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